CARBON SEQUESTRATION IN ALASKA'S BOREAL FOREST: PLANNING FOR RESILIENCE IN A CHANGING LANDSCAPE

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ABSTRACT

Northern ecosystems and those who rely upon them are facing a time of unprecedented rapid change. Global boreal forests will play an important role in the feedback loop between climate, ecosystems, and society. In this thesis, I examine forest carbon dynamics and the potential for carbon management in Interior boreal Alaska in three distinct frameworks, then analyze my results in the context of social-ecological resilience. In Chapter 1, I analyze comparative historical trends and current regulatory frameworks governing the use and management of boreal forests in Russia, Sweden, Canada, and Alaska, and assess indicators of socio-ecological sustainability in these regions. I conclude that low population density, limited fire suppression, and restricted economic expansion in Interior Alaska have resulted in a 21st-century landscape with less compromised human-ecosystem interactions than other regions. Relative wealth and a strong regulatory framework put Alaska in a position to manage for long-term objectives such as carbon sequestration. In Chapter 2 I model the landscape-level ecological possibilities for sequestration under three different climate scenarios and associated changes in fire and forest growth. My results indicate that Interior Alaska could act as either a weak carbon source or as a weak sink in the next hundred years, and that management for carbon credits via fire suppression would be inadvisable, given the associated uncertainty and risks. In Chapter 3 I perform a social, ecological, and economic analysis of the feasibility of switching from fossil fuels to wood energy in Interior Alaska villages. I demonstrate that this is a viable option with the potential benefits of providing lower-cost power, creating local employment, reducing the risk of catastrophic wildfire near human habitation, and earning marketable carbon credits. Finally, in Chapter 4, I assess how each of the above factors may impact social-ecological resilience. My results show some system characteristics that tend to bolster resilience and others that tend to increase vulnerability. I argue that in order to reduce vulnerability, management goals for Alaska's boreal forest must be long-term, flexible, cooperative, and locally integrated.

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INTRODUCTION

The global carbon budget has become a subject of interest across a broad range of disciplines in recent years, as the role of atmospheric carbon in global warming has become apparent. Natural resource managers, policy-makers and politicians have been pushing the academic community for answers to basic questions about the role of anthropogenic carbon; the ability of natural systems to ameliorate human impacts; and the potential positive or negative feedback loops that will govern climate change ecosystem interactions (Chapin et al. 2003). Increasingly, the ability of forest ecosystems to sequester carbon or to provide energy alternatives to fossil fuels has become of economic interest to states and nations as a broader range of extractive and non-extractive forest values become recognized and as nascent carbon-credit trading systems gain momentum. Thus, it is a matter of both global and local significance that the biomass and soils of Alaska's boreal forest contain large quantities of organic carbon, ecologically governed by a complex interplay between temperature, moisture, fire cycles, and human behavior -- all of which are subject to modification under changing climate regimes.

For several reasons, Interior Alaska provides a crucial perspective in the investigation of carbon sequestration potential and forest carbon dynamics First, boreal forests play an important and not yet fully understood role in global carbon budgets and global climate change; it is estimated that 40% of the world's reactive soil carbon is contained in high-latitude ecosystems (McGuire et al. 2000). Second, because the effects of climate change are demonstrably greater at higher latitudes, Alaska provides an early case study of the complex effects of changing climate on ecosystem health and resilience, and on the feedback loops between climate and carbon storage. Finally, Interior Alaska is currently facing – in addition to the pressure of ongoing climate change -- the simultaneous pressures of widening resource development infrastructure, globalization of markets, and stricter environmental policy.

Understanding and predicting the full range of variables that affect forest growth, use, and management -- as well as the feedbacks implicit to the system -- is crucial to carbon management, and forest management decisions in general. These variables include not only timber and non-timber extractive resources, but also ecosystem services such as watershed protection, cultural traditions, recreation, and aesthetics. Thus, models that incorporate biological, economic, and sociopolitical variables are necessary to address the full range of possible opportunities and outcomes.

Despite the inherent challenges of interdisciplinary work, there are several important arguments in favor of the approach that has been taken in this study. Single-discipline studies, particularly in the sciences, tend to focus on the ideal of complete information. However, it is generally impossible to have information that is both complete and broad, especially in a short time-frame. Increasingly, adaptive management systems in which new choices can be made as new information becomes available are proving to be more effective than less flexible approaches. Thus, scientists cannot rely unquestioningly on single disciplinary paradigms when dealing with real-world questions (Huber 1992). Interdisciplinary studies offer a means of addressing real-world problems that cross the boundaries between traditional fields. Such studies can broaden academic knowledge and creativity, breach communication gaps, and lead to greater social rationality and justice (Nissani 1997).

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CHAPTER 1

FOREST MANAGEMENT IN A NEW ERA OF CARBON ACCOUNTING: A CIRCUMBOREAL COMPARISON OF HISTORICAL AND CURRENT CONSTRAINTS

ABSTRACT

The simultaneous pressures of ongoing global climate change, widening resource development infrastructure, globalization of markets, stricter environmental policy, and development of carbon credit markets necessitate the adoption of a forest management paradigm that takes into account multiple cost-benefit streams at different levels of governance. I contend that the ability to successfully mesh local needs, regional economies, and global carbon accounting will depend not only on the strength of existing social, political, and economic infrastructure, but also on regional history. To support this assertion, I analyze historical trends in the use and management of the world's northernmost forests in four sociopolitical regions: northern Russia, Sweden, northern Canada, and Interior Alaska. My analysis reveals that historically, forest use has typically evolved from open access to sequential establishment of exclusive control, maximum sustained yield harvesting, and ecosystem management for multiple use. This series of transitions has often been linked to shifts in dominant economic and political paradigms, as internal and external sovereignty evolved, and as economic thought expanded from Classical definitions based on labor to neoclassical supply and demand and finally to the development of environmental economic thought. In most cases significant ecological and cultural damage occurred at each stage. In contrast, my results indicate that the forests of Interior Alaska are gaining almost simultaneous recognition for their multiple functions at the local, regional, and global scales, and have not undergone a prolonged series of management shifts or a pronounced cycle of degrading intervals. Thus, Interior Alaska provides a case study for how synchronous development of pressures at multiple scales may facilitate the transition towards a new management paradigm in a carbon-conscious world.

INTRODUCTION

Effective forest management is a difficult balancing act. Ecosystem dynamics are based on a myriad of variables, many of which are difficult to parameterize and predict, particularly at the temporal and spatial scales necessary for accurate modeling (Linder et al 2002). Managers must take into account the long-term and short-term needs of humans dependent on forest ecosystems – both directly and indirectly -- as well as the complex biotic and abiotic dynamics that occur within the ecosystem itself (Mangun and Mangun 1993). Human needs often conflict with one another and must be balanced accordingly, and yet many are inadequately accounted for using standard economic metrics (Daily 1997; Janssen 1998). Moreover, forests play important roles in the global climate system, as human-induced changes increasingly affect ecosystems at local and global scales (Melillo et al. 1996; Chapin et al 2004). This may be particularly true in the far north, where the feedback between climate change and ecosystem function is most pronounced (Keyser et al. 2000; Dargaville et al. 2002).

Boreal regions, including the boreal forests of Interior Alaska, are currently under intense scientific and political scrutiny regarding timber, oil, and natural gas potential; fire dynamics; habitat potential for game species; and wildland conservation. In addition, climate-change impacts and carbon sequestration potential are now taking their place in the policy limelight.

In response to the threat of global climate change, governing bodies and land managers are attempting to create an economic, social, and legal infrastructure to reduce carbon emissions. While the targets set for greenhouse gas reduction were the result of politically-charged negotiations, the dominant approach to achieving reduction, as expressed by the language of the Kyoto Protocol and subsequent agreements at Bonn and Marrakech, favors market-driven incentives (UN 1997; Schulze et al. 2002). Sequestration of carbon by the world's forests -- through afforestation, reforestation, or changes in forest management -- offers nation-states or polluting corporations the opportunity to offset their emissions (Wilman and Mahendrarajah 2002; van Kooten et al. 2004). Reducing fossil-fuel use by switching to renewable fuels, including biomass, provides another means of gaining carbon credits. Although the United States is not signatory to the Kyoto Protocol, voluntary market-based routes for obtaining sequestration credits have already been developed along parallel lines (CCX 2006).

In the world's boreal forests, management for carbon sequestration represents a new challenge for land managers and governing bodies, and necessitates the development of a more inclusive framework in which to consider the interplay of forest resources, local communities, and functioning ecosystems. It is likely that this challenge will be broached more successfully in some regions than in others, based on past, current, and future sociopolitical and ecological variables. Thus, carbon management offers an opportunity for Interior Alaska to embrace a management perspective that includes the broadest possible geographic and temporal range, and that focuses on the overall system rather than site-specific conservation.

In summary, I explore the following hypotheses in this chapter:

1. Changes in sovereignty and the development of environmental economics have heralded progressive shifts towards boreal forest management that encompasses a longer timeframe and a wider social, ecological, and geographic perspective.
2. Forest carbon management represents a logical extreme in the broadening of social, economic, and ecological interests.
3. Alaska's relatively strong social and political infrastructure and the compressed trajectory under which Alaska's boreal forest management has evolved may facilitate a transition to the above management framework.

Boreal Ecology and Resources

Interior Alaska's forest management options must be viewed in both local and global contexts, because the boreal forest plays a role in global carbon dynamics, provides ecosystem services both locally and globally, and is sensitive to anthropogenic change at multiple spatial scales. As outlined below, the social and ecological roles of this biome are shaped by geography, climate, fire, and human uses.

Alaska contains approximately 15% of the global boreal forest, which, in turn, comprises a third of the world's forest cover and eleven percent of the earth's surface (Burton et al 2003). The boreal forest in Alaska stretches across most of the state's Interior, and continues into Canada's Yukon. It covers roughly 52 million hectares, or about half of the statewide land area (Yarie and Billings 2002; Forbes et al 2004). Climatic change after the end of the last ice age led to thousands of years of gradually shifting species across Alaska, but the modern distribution of forest communities in Interior Alaska has been relatively stable for the past 4000 to 6000 years (Anderson and Brubaker 2004; Lloyd et al. 2006).

Boreal forests accumulate carbon in great quantities and for long durations, primarily in their soils (Dixon et al.1994; Hom 2003). Soils are acidic, slow to decompose, and often waterlogged. Low temperatures, permafrost, acidity, and in some cases low soil nitrogen lead to soil carbon storage on a scale not found in temperate or tropical forests (Dixon et al.1994, Billings et al.1998; Hobbie et al. 2002). High-latitude ecosystems contain about 40% of the world's reactive soil carbon (McGuire et al. 2000), with about 4.9kg/m^2 of carbon stored in biomass, and a much larger quantity -- 19.5 kg/m^2 -- sequestered in dead biomass, soil organic matter (SOM), and mineral soil (Kasischke et al 1995). Understory species -- including bryophytes, lichens, and woody and herbaceous vascular plants – moderate microclimate and permafrost, primarily by insulating soils (Bonan 1991; Zhuang et al. 2002). However, climate change is likely to alter nutrient cycling in boreal soils (Robinson 2002; Hobbie et al. 2002). This may occur either directly, through warming and drying, or indirectly, through altered fire cycles (Chapin et al. 2000; McGuire et al. 2002).

Fire is the primary natural means of disturbance and regeneration throughout much of the boreal zone (Viereck 1973; Van Cleve et al. 1991). Fires release a pulse of carbon from forest biomass, soil, and litter. In Alaska, highly flammable black spruce stands dominate poorly drained lowlands and cold upland sites (Rupp et al 2002; Viereck and Johnston 1990). The reproductive biology of black spruce is well adapted to fire; the heat opens semiserotinous cones, which release seeds onto burned-over charred organic and mineral soil. Aspen and birch sprout dense new shoots from surviving roots and stems, respectively, after fires. Black spruce bogs and forests cover approximately 44% of Interior Alaska and burn every 30-100 years, on average (Kasische et al. 1995; Rupp et al.2002; Yarie and Billings 2002). Larger white spruce and hardwoods occur along river valleys and on warmer south-facing slopes (Magoun and Dean 2000) and generally have much longer fire return intervals (Yarie 1981). Large wildfires impact not only local communities, but also state and national firefighting budgets; land management and development policies; and the global carbon cycle (Chapin et al. 2003). Fire disturbance is highly episodic, being widespread only in unusually dry years, thus increasing the challenge of managing a landscape for naturalness at multiple scales (Boychuk and Perera 1997; Johnson et al.1998). Feedback loops between climate, forest ecology, and fire frequency and intensity have the potential to alter the long-term carbon balance at the landscape level (Stocks et al.1998; Eugster et al. 2000; Chapin et al. 2000; McGuire et al. 2002; Hinzman et al. 2003).

Boreal ecosystems yield marketable resources that are significant both locally and globally. Although the trees are relatively small and slow-growing, greater worldwide demand for wood and fiber coupled with more fluid trade are putting those portions of the boreal forest that are accessible to global markets under increasing logging pressure. Likewise, oil and gas exploration is rapidly expanding in northern regions. Non-timber products such as mushrooms, birch syrup, and indigenous handiwork are important locally, and are starting to reach export markets. Tourism, based on ecological ideals such as the conservation of wilderness and the existence of healthy wildlife populations, is increasing (Burton et al. 2003). Carbon sequestration credits are also taking their place as a marketable resource within the existing economic matrix.

Ecosystem services provided by the boreal forest biome are crucial to the ecological and social resilience of the system (Teitelbaum et al. 2003). The boreal forest is a habitat mosaic of rivers, lakes, wetlands, and varied forest stand ages and types that supply marketable resources such as timber, oil, natural gas, and minerals; substantial subsistence resources, including fish, wildlife, berries, medicinal plants, and materials for construction and crafts; and less quantifiable non-market ecosystem services, e.g. biological diversity, watershed protection, and cultural and spiritual importance. Carbon sequestration is now shifting from being an uncounted ecosystem service to becoming a marketable resource.

Current large-scale industries in the world's northern forests include oil and gas extraction, mining, logging, commercial hunting and fishing, and the construction of hydro-electric projects. In boreal Alaska, hard-rock mining is an established and expanding industry, logging occurs on a relatively small scale, and oil and natural gas exploration may lead to the development of new extractive industry in the near future. These industries, while limited in area, are associated with significant road-building and other infrastructural development. Timber sales are primarily managed by the state, with 2005 receipts of approximately $64,000 from the Fairbanks area of the Tanana Valley State Forest, and no more than $500,000 annually from the Interior as a whole (Hanson pers. comm. 2006). By contrast, the Usibelli Coal Mine annually extracts approximately 1.1 million metric tons of coal with a value of roughly $50 million from 13,150 ha of leased State land (Usibelli 2004; Milkowski 2006). Currently, one large gold mine, Fort Knox, is in operation, and another, Pogo, is in the final stages of development. Fort Knox Mine is expected to produce about five million ounces of gold over a lifespan of 11 years, while Pogo Mine is expected to produce about five million ounces over 12-14 years, for a total estimated value of $3-6 billion (ADNR 2006). Smaller operations, while culturally significant, are economically dwarfed by these large-scale operations.

Hunting, fishing, and trapping are culturally and economically significant throughout the world's boreal forests, and moose, salmon, and caribou are important food sources for many residents of Interior Alaska. Although subsistence foods cannot legally be sold, some income is also derived from guided hunting and fishing trips. The annual harvest of subsistence foods in this region totals approximately 5.3 million pounds, or 54 pounds per capita. However, in rural parts of the Interior – which include all regions outside of the Fairbanks North Star Borough, the Denali Borough, and the Southeast Fairbanks Census Area -- the average is 454 pounds per capita, with 41% of caloric requirements and 293% of protein requirements being met by subsistence foods (ADF&G 2006). At a replacement value of $5/pound, this resource has an estimated value of $26.5 million annually, 50- to 100-fold greater than the value of timber harvest from the region (ADF&G 1998).

Many boreal species, including those important to subsistence users, depend on both the vastness of the landscape and the variability provided by natural fire cycles. Some species migrate over large distances: northern forests are intimately connected with other regions of the globe by hundreds of migratory bird species that summer in the far north (Irwin 2001). Large browsing and grazing mammals including caribou, wood bison, and moose inhabit boreal forests, as do numerous smaller mammals including beaver, wolverine, fisher, pine martin, mink, ermine, sable, snowshoe hare, red squirrel, lemming, and vole. Each species occupies a niche defined not only by the species it prefers as browse and cover, but also optimal landscape patchiness and time post-fire. The boreal zone also supports large omnivores and carnivores such as black bears, brown bears, wolves, Siberian tigers, coyotes, and lynx, although not all species are present in all regions. Predators tend to require large home ranges and are often sensitive to anthropogenic change. Wildlife also has cultural, spiritual, aesthetic, and scientific value – as do the full boreal biota and the landscape itself. There are over 2300 identified species in the North American boreal biome (Zasada et al. 1997), and although this represents relatively low biodiversity when compared to tropical regions, the range and phenotypic plasticity of each species tends to be broad.

Historical Boreal Forest Use and Management

Typical Evolution of Forest Management

Earlier summaries of boreal forest use and management (Kimmins 1991; Alverson et al. 1994; Östlund and Zackrissen 2000; Burton et al. 2003; CIF 2005) variously describe historical stages progressing from unregulated access through exploitation, regulated forestry, sustained yield, and finally "social" forestry. In this chapter I define a framework based upon four primary historical stages (Figure 1.1), place each boreal region in context socially, politically, and economically within it, and then expand it to include a final stage that incorporates carbon sequestration and other global concerns.

I demonstrate that regional differences in the time scale and synchronicity of these stages can be linked with local population density and associated pressure on forest resources, as well as with shifts in the dominant mode of economic and political thought. These stages are not entirely discrete. Lag and overlap occur between stages, and aspects of different management schemes persist, coexist and compete. The ideological, political and economic interests, institutions and relationships that support each stage persist or reappear; and each stage establishes its own set of persistent expectations and ideals about forest resource management. I contend that because the management trajectory has differed among regions of the boreal forest, comparisons among regions may elucidate the drivers of change and the social, economic, and ecological barriers to adoption of more holistic and more optimal management methods (Table 1.1). I further contend that some of the overlap between stages is the result of insufficiencies or inefficiencies in implementation of management regimes – especially in remote areas. These regional differences might also help define the challenges each region will face in incorporating carbon flux management into planning and conservation efforts.

I posit that achieving a forest management paradigm that fosters cultural, ecological, and economic resilience will prove most challenging in areas where either: 1) some aspect of this balance has already been compromised due to moving through the above trajectory over a protracted time period; or 2) existing social, political, and economic infrastructure seem unlikely to be able to support a shift to a more inclusive approach without a future series of management upheavals. Conversely, I predict that the greatest opportunity for fostering a resilient system will occur where both past and future trajectories are compressed.

This prediction runs counter to standard Democratic theory, which teaches that gradual incremental change -- rather than more abrupt transformation -- results in social inclusiveness and reduced conflict. Moreover, the Environmental Kuznets Curve (EKC) has often been held up as an unavoidable development trajectory, and proponents argue that industrializing regions must go through a phase of destructive environmental practices before increased per capita wealth allows for conservation (Ehrhardt-Martinez et al. 2002; Cole 2003; Stern 2004). However, I suggest that this pattern can be altered by political decisions and influenced by geography. In this case, Alaska has benefited from being among the newest and remotest regions of the United States, since less remote portions of the country have borne greater costs during the transition towards conservation.

Moreover, I contend that socio-ecological systems that have not been subject to a long series of transitions will ultimately benefit from an "advantage of backwardness." While more often linked to technological leapfrogging, such advantages can also be realized in systems in which "backwardness" is linked with low population densities and reduced land-use impacts (Nolan and Lenski 1985). In such cases, benefits are gained through observing the experiences of other areas suffering from past mistakes, entrenched interests, or unresponsive institutions. This learning is facilitated by social, political and cultural institutions that allow free flows of information and accommodate contending approaches. Development pressures and national and global attention have converged on AK boreal forests and forest managers (current and potential) during the historical period in which sustainable development and ecosystem management have become the rhetorical standards of environmental policy.

Stage 1: From Open Access to Exclusive Control

The earliest forest use took place in classic open-access situations, with forests treated as common-pool resources (Ostrom 1990). Traditional peoples -- including Native Alaskans and Siberians, the Sámi of Scandinavia, and the First Nations of Canada -- have been hunting and gathering in the boreal region for approximately six to ten thousand years, since the last glacial retreat (Burton et al. 2003). Throughout that time, they have altered the forest through localized burning and use of small amounts of wood for fuel and for construction (Lewis and Fergusen 1988; Natcher 2004). Traditional peoples have utilized non-timber forest products, including bark, roots, a wide variety of herbaceous plants and shrubs, and local wildlife (Bombay 1999). Although groups of people generally had known territories or ranges, the boundaries of these areas were blurred and no formal forest ownership existed for the vast majority of the past ten millennia. Thus, the political framework that most closely coincides with the open-access stage can be termed pre-sovereignty, where internal sovereignty is defined as either the legal right or the actual ability of the state to control people and processes within its borders (Litfin 1998).

Forest use during this phase can generally be characterized as stemming from local short-term needs (Figure 1.2). In other words, the forest user met immediate needs such as firewood, food, housing, and personal protection. He or she viewed the forest at the level of the stand, or several stands within an area that can be accessed on foot. Of course, the combined impacts of many forest users were neither exclusively local nor necessarily short-term.

For centuries, population density remained low enough that the "tragedy of the commons" (Hardin 1968) did not occur. However, as populations grew, competition between groups and tribes in some areas became more frequent or more intense, and common resources became noticeably depleted. Increasingly, defined ownership and control became established, including individual tenure, group ownership, and government claims. However, the time at which this shift took place and the amount of ecosystem degradation that occurred prior to defined ownership vary widely, as will be seen below.

By far the earliest historical shifts in the boreal forest took place in Scotland, which was once covered by the ancient Caledonian forest (Birks 1988; Tipping 1994). Early Celts harvested wood to build stockades and clear land for agriculture (Macklin et al. 2000). In the Middle Ages the land transitioned from common property to formal custody in the form of sovereign feudal control. With the establishment of ownership came efforts to curb over-exploitation. Some of the earliest written laws – dating from the 12^{th} century – related to allocation and protection of forests and forest resources;

however, these efforts were too little too late (Smout 1997). At present, less than 2% of native forests remain, meaning that Scotland is rarely counted at all when boreal forest management is discussed (Angelstam 1998). However, it represents an interesting antithesis to the case in Interior Alaska.

In Sweden, forest privatization generally predated or coincided with over-use. Lands gradually became formally owned either by families or by the government, from the Middle Ages onwards. By the end of the 18^{th} century a small percentage of forest land was cleared for agriculture, and farmers sought potash, tar, and timber from nearby forests (Axelsson 2000). Population density grew throughout the 1880's, and exploitation also increased. By the 1850's, significant logging was occurring, and in the 1880's the Swedish Forest Service began offering contracts to sawmills (Östlund et al.1997). Formal management was solidified with the passage of the 1903 Forestry Act (Östlund and Zackrissen 2000).

Human impacts at the landscape level in the Russian boreal were minimal prior to the establishment of exclusive control of forest resources. During the Imperial period (1682-1917) ownership and management were essentially feudal, with some localized land conversion for agriculture. However, Russia contains roughly 60% of the global boreal forest (Forbes et al. 2004), and local impacts were minor compared to the vast expanse of Siberia. The first significant exploitation in this region was during the Soviet era, starting in 1917. At this point, forests – and indeed all land – had been officially claimed by the Federal government, and official policy called for rapid population growth and development in the boreal (Forbes et al 2004).

Boreal Canada is rich in forests, and First Nations peoples have utilized forest resources for millennia. In some cases they may have ignited forest fires for a range of uses including hunting, pest management, and creation of game habitat (Williams 2001; Natcher 2004). Although short-term effects and local impacts may have been great, they were not qualitatively different from the effects of lightning-caused fires, which probably accounted for most of the area burned and constituted a disturbance regime to which boreal species were well adapted. Consequently, the system appears to have been relatively resilient to these anthropogenic fires (Lewis and Fergusen 1988; Natcher 2004). However, depletion of desirable softwoods in temperate zones and increased demand pushed logging north in the mid-1800s (Reed 2001). In Canada, official ownership of previously unclaimed forests occurred as a side-effect of Confederation, via the British North America Act (BNA) of 1867. The BNA granted all timber resources (as well as minerals and inland fisheries) to provincial governments (Reed 2001). Formal management arrived soon afterwards (CFA 2005).

Forest use in Interior Alaska prior to1800 roughly paralleled that seen in boreal Canada, but it did not follow the same trajectory thereafter. Alaska was officially a Russian territory prior to 1867 and U.S. territory afterwards, but the Non-Native population was extremely small until the gold rush, circa 1900 (Naske and Slotnik 1987). The steamboats and dredges used by prospecting communities had enormous wood-burning capacities, but almost all timber harvesting took place along rivers, within a few hundred feet of the water (Roessler 1997; Magoun and Dean 2000). Accidental ignitions also probably increased; however, human-caused fires are still estimated to account for only 10% of the total area burned (Murphy et al. 2000; DeWilde and Chapin in press). In the latter half of the twentieth century, local wood processors started business, and as population increased, more cabins were built and more firewood used, but this type of use was small-scale and localized. Land ownership in much of the boreal zone was undefined prior to statehood in 1959. Even after statehood, large land areas remained in ownership limbo; they were doled out to Federal, State, and Native land managers under the Alaska Statehood Act, the Alaska Native Claims Settlement Act (ANCSA 1971) and the Alaska National Interest Lands Conservation Act (ANILCA 1980) (Naske and Slotnik 1987), but some of these land transfers have still not been completed. Currently, Alaska's boreal forest is primarily on state and federal land, with a mosaic of private and Native holdings. However, a high percentage of the most productive forest is state-owned, due to the selection process allowed under the Alaska Statehood Act.

Throughout the world's northern forests, direct forest ownership or exclusive control of forest resources coincided with changes not only in human behavior in relation to the forest, but also in political frameworks, economic paradigms, social norms, and available knowledge. In most cases, governance over the boreal forest was established as part of the establishment or expansion of internal sovereignty within existing states. Forest claims in northern regions were generally established following the rise of Classical economic thought, as first described by Adam Smith (1776), and prior to the rise of neoclassical economics in the late 1800s and Keynesian macroeconomics in the early to mid 1900s. Classical economics identified the wealth of a nation as the product of labor applied to the land. Marxian economics, which prevailed in Russia during the Soviet phase of command-and-control forest management, was also based on a labor theory of value. With land ownership – either private or state -- came a belief among the dominant culture (which often invaded or displaced the indigenous culture) that all land should be controlled and altered by humans for their own benefit. In addition, the concept of the 'invisible hand" introduced the idea that individuals, through pursuit of their own wants, will tend to inadvertently work for the good of the community as a whole (Smith 1776). Under Classical and early neoclassical modes of thought, land and capital were distinct; thus no concept of natural capital existed. Economically speaking, ecosystem services did not exist.

Stage 2: From Exclusive Control to Maximum Sustained Yield

The development of a maximum sustained yield management philosophy within the context of exclusive control often occurred as a gradual progression rather than as a discrete shift, as governments struggled with broader socio-economic and political challenges. These included increased population pressures, growing demands from an urbanizing society, and the development of government-assisted agricultural and industrial development strategies.

Contrary to the predictions of Classical economic theory, defined ownership and/or exclusive control of forest resources frequently resulted in sub-optimal use patterns. Governance often arises when human needs begin to impinge upon one another (Young 1997); thus, the withdrawal of boreal forests from the commons was generally instituted in places and times when population pressure and development pressure made over-exploitation inevitable. However, if the hegemonic paradigm of the time was harnessing natural resources in the service of growth, industrialization, prosperity, power, and sovereignty, problems often occurred because only a small subset of forest products and services were considered.

In contrast to early forest users, who had considered costs and benefits that were personal or communal and local, newly appointed forest management entities generally expanded the geographic focus of both potential harvest and potential markets to include a larger region -- or even a whole nation-state. However, they maintained a predominantly short-term perspective that left no room for considerations as lengthy as ecological succession and forest regrowth (Figure 1.2), and thus did not curtail the tendency to over-exploit.

The subsequent gradual shift to management for maximum sustained yield (MSY) was representative of early philosophies of conservationist management, which focused on a longer-term stream of benefits but maintained a narrow focus on what type of benefits to consider. Under MSY, the annual allowable cut (volume of wood harvested) does not exceed the annual increment (volume of new wood grown). Forests are generally harvested on a rotation system, regardless of whether this accurately mimics natural processes. Managers strive to increase the annual increment through thinning, planting, introduction of more productive species, eradicating competing species, and scheduled harvesting.

The early application of MSY was based on both laws and regulations (command-and-control mechanisms) and on market mechanisms (Irwin 2001). It was affected by the tenets of neoclassical economics, which first appeared in the late 19th century, as well as by state-sponsored rapid industrialization. These two influences were often at least partly in conflict with one another.

The neoclassical school of thought attributes value based on supply and demand, rather than on labor, as in Classical economics. Application of Classical thought tends to support immediate exploitation of resources, since adding more labor (e.g. more harvesting of timber) increases total wealth. In contrast, Neoclassical thought allows for the substitutability of labor, land, and capital, and considers the present and future tradeoffs (opportunity costs) associated with a particular use of resources. Neoclassical thought further suggests that the privatization of all resources and the free market exchange of any goods and services derived from those resources will maximize total societal benefit. However, this assertion presupposes no market failures. In other words, one must assume that individuals are rational actors who will act in their own best interests; that individuals have complete information about the costs and benefits of their actions; and that all costs and benefits accrue directly to the individual users without either positive or negative externalities (defined as uncounted benefits or costs, respectively, to society at large). Future costs and benefits are accounted for, but at some discounted rate compared to current cost and benefits.

Because market failures do occur in the real world, government manipulation of markets or command-and-control strategies may be needed to make the market function more efficiently. In reality, however, such manipulations are often made for other reasons, such as spurring immediate economic growth. Rapid industrialization was state-sponsored rather than truly market-driven, and future costs and benefits were often given little consideration in comparison to immediate gains. Although MSY introduced an element of conservation into resource use strategies, it sought to conserve only those attributes that fostered and supported the goals of industrialization: fiber and timber, but not cultural or environmental qualities.

The advantages of MSY are that it can be calculated with relative ease, it provides a fairly steady and measurable benefit stream, it affords a sense of long-term guardianship, and it can be made to conform to fairly inflexible and static systems of laws and regulations. The downsides, however, are many. Even if loss of wood to fire, insects, and other natural disturbances is taken into account, MSY generally does not account for the benefits provided by forests for wildlife, climate regulation, species diversity, maintenance of soil fertility, genetic diversity within species, tree-age diversity, or fragmentation and patchiness of the landscape. Thus, it tends to result in losses within each of these categories.

Scandinavia was the first boreal region to experience these patterns of shifting societal norms and associated loss during this phase. In Sweden, commercial timber harvesting began in the 17th and 18th centuries, and by 1900 large tracts were contracted to pulp and paper companies. At first, these contracts were demand-based rather than supply-based; there was little if any expectation that the forest would regenerate or that timber supplies would be sustainable (Burton et al. 2003). By the 1920's, fire had been virtually excluded as a major disturbance (Axelsson 2000). Non-timber ecosystem services and indigenous rights were not recognized, and native deciduous species were removed in favor of plantations of Scots pine (Östlund and Zackrisson 2000). As ideas of future economic value and conservation took hold, MSY became the dominant mode of operation. However, aggressive fertilization, replanting, and other intensive silvicultural methods resulted in lucrative yields of wood and pulp, but reduced biodiversity (Östlund et al. 1997; Nordlind and Östlund 2003). As much as three quarters of Sweden's eight thousand invertebrate species may have been lost due to reduced forest complexity and habitat destruction (Gawthrop 1999). Intensive management resulted in species shifts, single-aged single-species stands, and very few snags, older stands, or wetlands (Nordlind and Östlund 2003).

In Russia during the Soviet era, the focus within forestry was on meeting production quotas, irrespective of efficiency of technique or sustainability of ecosystems (Gawthrop 1999). Soviet land use strategies and policies can only be understood in the context of state-controlled rapid industrialization policies aimed at catching and surpassing the West in a very short timeframe. When the first Five-Year Plan was launched in 1928, the economy was 30-50 years behind the most advanced industrialized nations; Stalin's goal was to catch up as rapidly as possible (Holzman 1982). Thus, replanting was often ineffectively performed, and clear-cut logging was focused on the most lucrative and accessible stands (Forbes et al 2004). Most government-run logging enterprises collapsed with the fall of the Soviet Union. However, in the 1990's a significant and mostly unregulated trade in raw logs grew up between Russian and Europe, as well as with Japan, China, and South Korea (Gawthrop 1999).

The Canadian government has traditionally been aggressive in its desire to extract wealth from its forests, and its policies regarding the disposal of public land in the late 19th and early 20th centuries emphasized rapid growth and development (Gates and Gates 1984). Formal forest management began when a Chief Inspector of Timber and Forestry was appointed under the Department of the Interior in 1899 (CFA 2005). From the 1930s onwards, control of most public lands shifted to the provinces, and at the same time, a conservation ethic gradually began to form (Gates and Gates 1984). However, sustainable yield wasn't addressed until the Forest Act of 1961 determined allowable cuts and established quotas. Even then, this system only considered wood production by volume, and ignored cultural and ecological functions of the forest. Canadian forestry practices led to the widespread creation of even-aged stands, none of which were older than the short rotation lengths preferred by the timber industry, and few of which were subject to natural fire disturbance (Bergeron et al. 1999, 2004).

Although the US purchased Alaska in 1867, the remote and sparsely populated Interior of the state was largely unaffected by the Western land management policies that epitomized this era, such as the Homestead Act of 1862, which gave away public land in order to catalyze its development (Gates 1963). Instead, formal forest management in Alaska's boreal zone wasn't initiated until the Alaska Forest Resources and Practices Act (AFRPA) was passed in 1978 (Alaska 2000), and the Tanana Valley State Forest (TVSF) was created 1983. The first management plans for the 15-million-acre Tanana Basin as a whole and the TVSF (representing almost 2 million acres within that area) were not completely until 1985 and 1988, respectively (ADNR 2001).

Although forest use – including timber harvesting – was essentially demand-based under the original TVSF management plan (ADNR 1983), the era in which it first took force was one in which conservation-oriented social, political, and economic paradigms had already gained some momentum. Not only did both the plan and the superceding AFRPA (Alaska 2000) include language that clearly recognized some of the non-extractive values of forests, but such ideals were also already commonly recognized, and actively supported by individuals and non-governmental organizations (NGOs) such as the Northern Alaska Environmental Center (founded in 1972), and later the Alaska Boreal Forest Council (founded in 1993) and the international Taiga Rescue Network (founded 1992) (NAEC 2005; ABFC 1995, 2005, TRN 2005). Thus, an increase in demand and the brief development of a round-log export market circa 1990 led to increasing public outcry and a demand for public education, agency accountability, and ecologically and socially sustainable management. By 2001, a revised management plan was produced based on maximum sustained yield, but also incorporating newer ideals of multiple-use ecosystem management, and incorporating extensive feedback from the scientific community and from the public (Tanana 2001; Dawe et al. 1994; Ruggles 1994, Community 1995, Dawe 2000). Because this transition took place at a time when newer, broader notions of forest management were already readily accessible and accepted, it was far briefer and less ecologically and socially deleterious than in other regions.

Stage 3: From MSY to Sustainable Ecosystem Management

In the past forty years, in response to changing attitudes and an awareness of increasing pressures that threaten the integrity of natural systems -- particularly in more developed nations -- the idea of MSY has given way to a more encompassing concept of long-term sustainability and multiple use of forests. Forests are recognized for their importance as sources of extractable resources and ecological habitats; their role in watershed conservation; and their aesthetic and recreational values. Sustainable ecosystem management ideology tends to include longer-term costs and benefits than previous management regimes (Figure 1.2), although it still concentrates on the local and regional level rather than incorporating the full range from local to global. The idea of sustainability has become iconic within many forest management programs, despite the difficulty of both defining and achieving it. By the 1990's, it was the preferred paradigm both nationally and at the international level (Burton et al. 2003).

In many northern forests, the transition from MSY management to management for ecosystem services mirrored the rise of environmental economics, a school of thought that focuses on addressing market failures that lead to environmental degradation. Such market failures occur when individuals make less than optimal choices because they lack options, because they do not possess full knowledge of all costs and benefits, or because they do not have to pay the true costs of their choices due to market externalities. Externalities tend to be a problem where public goods are concerned. Modern economics defines a "public good" as a good whose cost is indivisible, and for which it is not practical to charge the user. The atmospheric is an example of a common pool resource, and the services it provides are public goods, at least within our current level of international governance. Atmospheric carbon, in as much as it is associated with global warming, is a negative externality, and greenhouse gas reduction is a form of collective action that can contribute to the public good of a more stable climate. Like all collective actions that produce public goods is subject to free-riding and other collective action problems.

As the environmental movement has taken hold, and as environmental economics has been developed, attitudes have changed at a fundamental level. The idea of what a forest is has been revised from a simple definition based on preferred tree species to a complex ecosystem definition based on the full range of ecosystem services provided by forests. Changing management strategies in boreal forests have been driven by these changing attitudes (Simberloff 2001).

However, the transition has not been made without significant obstacles caused by the political, economic, and ecological remnants of past management. Anachronistic incentives such as governmental subsidies for resources extraction can hinder efforts towards sustainability. Many undercounted values of the ecosystem exist outside normal supply-and-demand liberal economic accounting systems; such values include subsistence foods, watershed values, carbon sequestration, and hard-to-measure intangibles such as quality of life and the existence value of wild areas and threatened species. Another problem associated with international, intercultural, or even domestic policies on sustainability is that "sustainable" is defined in different ways at different times and by different people. At issue is the timeframe within which a particular resource is expected to be conserved, as well as the level of ecological integrity that must remain.

It should be noted that considerable overlap between stages has often occurred, due in part to resistance to change by beneficiaries of the older regime. When these detractors hold significant political currency, transitions can become protracted, with newer approaches and ideas existing within the context of older management structures. This has been particularly true in the adoption of environmental standards that support sustainability.

Northern Europe is now transitioning towards a more culturally and ecologically sustainable pattern of boreal forest management, albeit unevenly and not always successfully. At the far end of the spectrum, British forest ecologists are struggling to maintain some subset or the historical Scottish native flora in small "semi-natural" plantations (Humphrey et al. 2000). Cultural ties to forests have been lost, and in some cases forest restoration is directly opposed by local people (Crumley 1997). In Sweden, although the Forestry Act of 1993 codified management for biodiversity and other ecological and social factors, much of the original complexity and biodiversity of these forests has already been lost, as have many of the cultural traditions associated with forest-dwelling peoples. Over 2000 Swedish forest species are currently on the IUCN Red List of threatened, endangered, or extinct species. Natural fire cycles were eradicated in the 18th and 19th centuries and have never been reestablished (Östlund et al. 1997; Niklasson and Drakenburg 2001).

Russia has had a particularly difficult time embracing sustainable forestry, in part because of structural problems in its government during the transition from communism to capitalism. Neoclassical economics and its environmental offshoots have not arisen naturally in Russia, and have not taken hold at the operational level (Carlsson and Lazdinis 2004). There have been problems determining ownership of forested lands, in clarifying which lands fall under local, state, or federal jurisdictions, and in financing forest conservation (Korovin 1995). Strong environmental laws relating to forest practices have been adopted on both a regional and national level in Russia (Burton et al. 2003), but they are rarely enforced (Forbes et al 2004). The Russian Principles of Forest Legislation were adopted in 1993. They contain a set of rules and guidelines intended to protect the forest during the transition to a market economy, to decentralize forest management, and to increase the power of federal and local authorities to enforce forest laws (Korovin 1995). However, Russia continues to serve as an unfortunate example of the inefficacy of a system that tries to address national and global-scale issues when local and immediate needs have not been met (Figure 1.2.) While Russia's historical forest management trajectory has placed fewer strains on the ecosystem than that in Scandinavia, its future trajectory may well replicate a similar series of damaging transitions.

Meanwhile, the Canadian Forest Act of 1961 was revised repeatedly in the 1960s and 1970s, but it was not until the nineties that ecosystem values and biodiversity were factored in (Henderson 1996). Approximately 65 percent of Canada's boreal forest is currently under long-term tenure leases to multi-national forest companies, mining interests, oil and gas exploration or hydro development. Although management goals now include mimicking natural disturbance when harvesting, there is currently little agreement as to how this can best be achieved (Haeussler and Kneeshaw 2003). It can be argued that the Canadian forestry sector, in contrast to the US, has not yet emerged from an operational framework in which short-term revenues from extractive industries take precedence over ecosystem values (Innes and Peterson 2001). Culturally, the picture is also problematic. About 80% of First Nations people live in Canada's forested regions, and rely on forests for subsistence and other traditional activities, but First Nations recently had to file suit to legally determine that management activities that curtail traditional Aboriginal activities (through fragmentation or loss of habitat) impede existing Aboriginal and treaty rights and that forestry companies have the obligation to exercise due diligence in order to ensure that Aboriginal rights are not infringed upon. Fire control practices may also pose a barrier to effective ecosystem management, although as yet there is little evidence that suppression has significantly altered landscape patterns (Johnson et al. 1998; 2003)

Boreal Alaska has thus far experienced fewer ecological, social, and economic losses than other boreal regions during its transition towards sustainable ecosystem management. This relative advantage stems from lack of a long history of poor management and from a strong regulatory framework, rather than merely from recent management choices oriented towards a more holistic perspective, such as those expressed in the AS 41.17 (2000) and the TVSF management plan (ADNR 2001). The regulatory framework provided by ANILCA has arguably done more to protect ecosystem integrity in the far north than any single policy in any other nation (Forbes et al. 2004). Its stipulations include ecosystem protection, provisions for human uses, and opportunities for scientific research (ANILCA 1980).

However, in the past decade, Alaska has merely matched, or in some cases lagged behind Canada and northern Europe in the development of concrete plans for sustainability. The recent policies of the Alaska state government have been strongly pro-extractive, and at times antagonistic to ecosystem-level thinking. The state legislature has recently passed laws to increase the timber, oil, and gas industries in the boreal, even when state land managers advise against the new rules and regulations. For example, in 2002, the State of Alaska passed a new statute that required Minto Flats State Game Refuge, an area of high value as wildlife habitat, to be open for oil and gas leasing (HRC 2002). In addition, Senate Bill 149, which became law in 2003, changed the primary purposes of Alaska State Forest management from 'multiple use' to 'timber management ... while allowing other uses'; it also limited the creation of new riparian protections, and eliminated consideration of certain impacts on fisheries, wildlife and other users (Taylor 2003).

In Interior Alaska, the impacts of natural fire still outweigh anthropogenic effects. As compared to Canadian forests, access is more difficult, total forest area is smaller, marketable forest stands are smaller and more dispersed, and the size and desirability of the trees available is generally lower (Wurtz et al. 2006). Thus, demand has been low except during market peaks for timber products. Moreover, for the most part, large fires have been allowed to burn unchecked in Alaska, despite official US policy that called for total suppression. A more pragmatic and ecologically sound policy formalized in 1991 now allows fires to remain unsuppressed in unpopulated areas (Chapin et al. 2003; Todd and Jewkes 2006).

Despite a state administrative policy focused on resource extraction, the trend within management agencies and at the grassroots level has been increasingly towards decision-making that takes into account diverse stakeholder input. In some cases, intermediate stages such as MSY have been leapfrogged. For example, the original 1983 version of the Tanana Valley State Forest Management Plan, which dictates the use of some of the region's most productive white spruce forest, was based on demand-based harvesting strategy without reference to forest productivity or measurable standards of ecosystem integrity. Revisions to the plan, however, were the result of years' worth of public meetings and input representing Native interests, environmental NGOs, logging and mining interests, and scientific research. The new plan allows for harvesting over much of the forest, but also specifies requirements relating to reforestation, habitat protection, and protection of traditional activities (ADNR 2001).

The system in place is still in transition, and is lacking in both the adaptive flexibility and the cross-scale collaborative strength necessary to address forest management in a global context and a multi-generational timeframe. However, low timber market pressures, relatively small populations, great national wealth, and strong infrastructure all accord a temporary advantage, which, in combination with historical advantage described above, augments the resilience of Alaska's boreal forest.

Conclusion

The past two decades have been a time of rapid social, ecological and political change with respect to management of the world's boreal forests. Large increases in demand for timber, the collapse of the Soviet Union, the globalization of business and industry, increased environmental concerns among the public, and the threat of global climate change have together caused an unprecedented revolution in attitudes, necessitating an equally dramatic change in management paradigms.

In many forested northern regions, management frameworks have undergone a protracted series of transitions which have gradually incorporated a broader spectrum of costs and benefits, as society has become aware of social-ecological feedbacks occurring at larger temporal and spatial scales. An evolutionary progression from open access to regulation and then to MSY has only recently been overlaid with imperfect attempts to manage forests at the ecosystem level. Historical management practices that only accounted for a temporally or spatially narrow set of costs and benefits have often led to negative economic, ecological and social impacts that may be difficult to reverse, even under strong regulatory frameworks. Where such frameworks are weak, the challenges are even more severe.

However, in Interior Alaska, the historical timeframe within which intensive human use and management has taken place is relatively short; the proportion of the forested landscape impacted by such use has been comparatively limited; and the existing political infrastructure is reasonably robust. Fire has not been completely suppressed, native species have not been eradicated, no large-scale timber industry has taken hold, and subsistence land uses are still maintained. Although significant alterations have taken place in each of these categories, and although road networks, mines, and increased human populations have taken their toll on forest integrity, the Alaskan boreal forest remains less altered than similar ecosystems elsewhere in the world.

As a result, Alaska is now faced with a compressed trajectory, in which new information about ecosystem dynamics, new ideas about conservation and management, more environmentally-oriented economic theories, new understanding of climate change and carbon dynamics, and new socio-political collaborative frameworks are available concurrently rather than piecemeal. It should be possible to quickly bypass less desirable stages of forest management in favor of creating a more socially, ecologically, and economically sound management strategy. Similar technological and socio-economic leapfrogging has sometimes occurred in developing nations; in Alaska, the opportunities are even greater, due to the ability to rely on the infrastructure of a wealthy developed nation.

The Kyoto Protocol and the US response to it indicate the political and economic complexities brought by globalization. Alaska's boreal forest management is at a cusp. Managers may flounder, making the same mistakes made in other regions and failing to integrate planning efforts between governing bodies, across spatial scales, and among disparate interest groups. A short-term economic outlook coupled with unbalanced political influence among stakeholders may result in management that is optimal for no one and that jeopardizes the long-term resilience of the system. On the other hand, if effective collaboration, adaptive management strategies, and a broader economic approach are applied, Interior Alaska may be among the first regions to effectively institute broad-based forest management that balances a wide range of interests in a changing landscape.

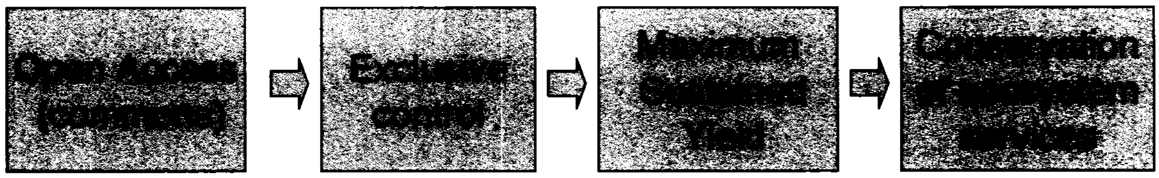

Figure 1.1. Typical stages of forest use and management. New scientific information and new social/cultural norms are associated with each stage and a reorganization phase occurs during the transition between stages.

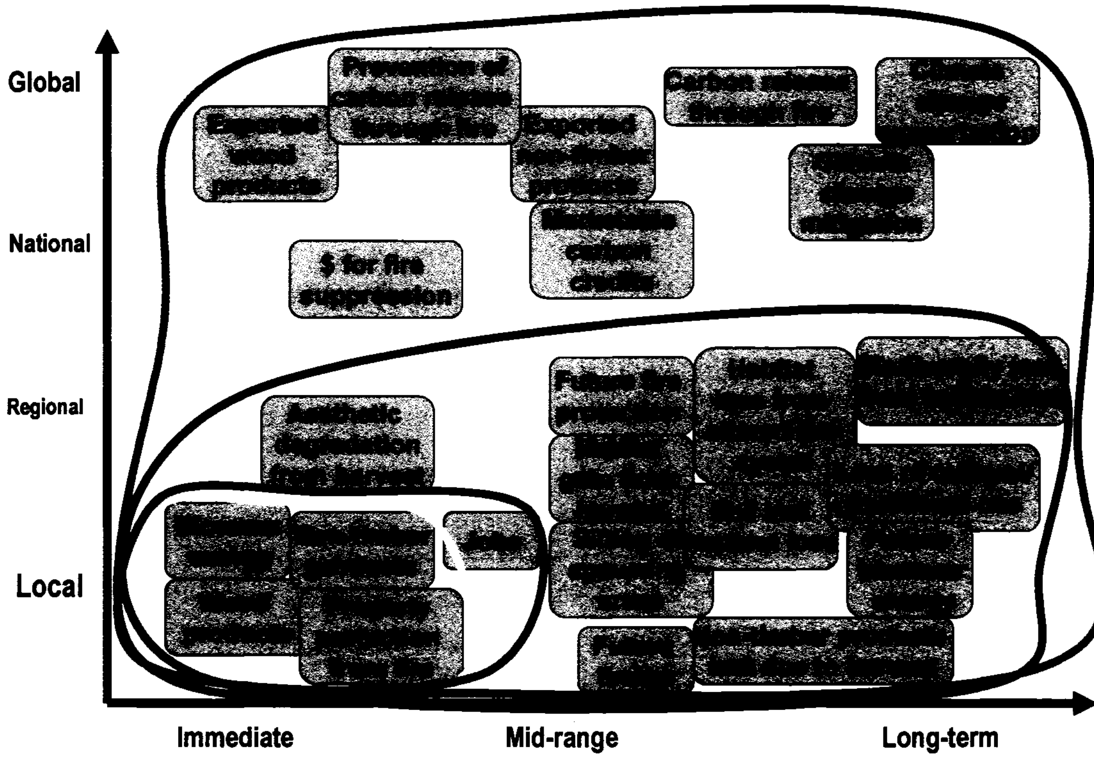

Figure 1.2. Costs and benefits by geographic and temporal scale. The graph depicts the tradeoffs of timber harvest and fire suppression, with benefits shown in green and costs in red. The smallest loop shows the range of concerns typically incorporated under common access or management prior to MSY. The next circle depicts MSY management concerns, the third extends to include the concerns expressed under management for conservation of ecosystem service, and the largest loop includes the full range of variables at stake, including carbon management.

Table 1.1. A comparison of forest use and management histories in four boreal regions. The trajectories from open access to more comprehensive management regimes took place at a very different rate in different parts of the boreal, with that in boreal Alaska being the most synchronous.

	Interior Alaska	Sweden	Canada	Russia
Open access problems occur	Some localized overexploitation during the Gold Rush era, 1900	Increasing uncontrolled harvest circa 1800-1900	Increased demand pushes forest activities north in the mid-1800s	Localized land conversion for agriculture, some feudal disputes during Imperial period (1682-1917)
Exclusive control established	Small private claims 1867-1959; Statehood, 1959; ANSCA, 1971; ANILCA, 1980	Gradual, from Middle Ages onwards	Private claims followed by Confederation (British North America Act) 1867	Gradual, with mixed state and private ownership as of 1917; 100% government ownership established during Soviet period
Formal forest management	Alaska Forest Resources and Practices Act, 1978; Tanana Valley State Forest created, 1983	1903 Forestry Act	Chief Inspector of Timber and Forestry appointed under the Department of the Interior 1899	The Basic Law on Forests 1918; The Forest Code of 1923
Fire prevention established	Alaska Fire Control Service assigned to suppressing wildfires 1939	Fire prevention increased as harvest increased; natural fire cycles eradicated circa late 1800s	Forest Fire Prevention Week established 1920	Traditional rural fire prevention centered around villages, but natural fire cycles never eradicated
Management for MSY	Tanana Valley State Forest management plan update 2001; Senate Bill 2003	Forestry Act of 1979	Forest Act of 1961	Division of forests into three groups circa 1943; development of MSY paradigm 1943-1993
Change to fire management	Alaska Wildland Fire Management plan 1991	Natural fire cycles never reestablished	Forest Fire Prevention Week renamed to National Forest Week 1958	Fire prevention is still the official priority, although more pragmatic methods are actually in use.
Management for biodiversity & ecological factors	Tanana Valley State Forest management plan update 2001	Forestry Act of 1993	First Canada Forest Accord signed at the 7th National Forest Congress	Russian Principles of Forest Legislation, 1993; Forest Code of 1997
Carbon management	Alaska Senate Bill 2004; not yet implemented	Sweden ratified the Kyoto Protocol in 2002	Canada ratified the Kyoto Protocol in 1998	Russia's ratification allowed the Kyoto Protocol to take effect in 2005

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CHAPTER 2

THE EFFECTS OF CLIMATE CHANGE ON ECOSYSTEM CARBON IN INTERIOR ALASKA BOREAL FORESTS: THE IMPRACTICALITY OF MANAGING FOR CARBON CREDITS VIA FIRE SUPPRESSION

ABSTRACT

Boreal forests play an important role in global carbon cycling. As the social, political, economic, and ecological importance of carbon accounting increases, predicting the response of Alaska's boreal forests to ongoing climate change and altered fire cycles becomes increasingly significant. To predict carbon dynamics in the boreal forests of Interior Alaska, I used a landscape-level model of carbon dynamics, the Carbon Budget Model for the Canadian Forest Sector (CBM-CFS3), to model changes in terrestrial carbon stocks over a period of 100 years under current climate conditions and under two different warming scenarios: +1.5°C and +4°C. Simulations were based on the implementation of CBM-CFS3 for the Boreal Cordillera Region of the Yukon Territory. However, model parameters, which included growth curves and cover type by species, were derived from data for forests in Interior Alaska. In order to simulate the effects of wildfire on the system, I used outputs from a spatially-explicit frame-based fire model, ALFRESCO, to create simulated fire cycles under each climate scenario. Results showed that the net carbon balance in boreal Alaska is likely to remain slightly positive under conditions of no climate change or warming of +1.5° C. However, if more extreme warming occurs, either slight increases or decreases in carbon are possible. Given this uncertainty; low levels of carbon gain or loss; and the current limitations of national and international carbon accounting; my analysis suggests that landscape-level fire management for carbon sequestration and the sale of tradable carbon credits is impractical at this time.

INTRODUCTION

Anthropogenic release of carbon via fossil fuel combustion and land use change is recognized to contribute to the increase in atmospheric levels of greenhouse gases, and triggering global climate change (Karl and Trenberth 2003; Karoly et al. 2003; Canadell et al. 2004). Nonetheless, there is still much uncertainty regarding the rate of global climate change, the quantitative roles of communities and ecosystems in exacerbating or mitigating this change, and the magnitude of positive or negative feedbacks between terrestrial ecosystems and the climate system (Kurz and Apps 1994; Chapin et al. 2000; Haque and Burton 2005)

Global boreal forests are likely to play a particularly important role in future global carbon balance (McGuire and Chapin 2006), because they contain a large carbon stocks, especially in soils, and are situated in a region of pronounced warming (Price and Apps 1995; Bonan 1995; IPCC 2001a, 2001b). It is currently unclear whether boreal forests and other high-latitude systems are acting as an overall source or sink for atmospheric carbon (Chapin et al. 2000; Clein et al. 2002). Although carbon accumulation by boreal forests may account for half or more of the "missing sink" – previously unaccounted carbon being removed from the atmosphere (Mahli et al 1999; Liski et al. 2003), this role could be reversed if current warming and drying continue (Kurz and Apps 1995; Serreze et al. 2000; Harden et al. 2000).

Changes in disturbance regime could also contribute to changes in boreal carbon storage. Wildfires release enormous quantities of carbon that are taken up again only gradually over the course of forest regrowth (Amiro et al. 1999). Thus, changes in the pattern or frequency of fire could alter the overall carbon balance (Kurz et al. 1997; Harden et al. 2000). Also, increased human presence and economic interests cause shifts in boreal cover type, stocking, and fire (Kurz and Apps 1995). Finally, thawing of permafrost can create new landscapes through subsidence or drainage (Jorgenson and Osterkamp 2005) that can greatly alter soil carbon transformation associated with decomposition and methanogenesis, possibly enhancing the release of CO_2 and CH_4 (McGuire and Chapin 2006; McGuire et al. 2006; Zhuang et al. 2006). Thus, it is important to assess how carbon storage in the boreal forest will respond to potential climate change.

Interior Alaska Forest Ecology

Interior Alaska is situated north of the Alaska Range, south of the Brooks Range, west of the Canadian border, and east of the ocean-moderated Bering Sea. Its continental climate is characterized by short warm summers, long cold winters, and low precipitation. Soils are generally weakly developed with discontinuous permafrost varying according to slope, aspect, elevation, and latitude (Slaughter and Viereck 1986; Ping et al. 2006).

The majority of Interior Alaska is forested, with the exceptions being high-elevation tundra, rivers, lakes, and muskegs. The major forest cover types in Interior Alaska are black spruce (Picea mariana [Mill.] B.S.P.), white spruce (*Picea glauca [Moench] Voss*), Alaska birch (*Betula neoalaskana Marsh.*) and aspen (*Populus tremuloides Michx.*), with balsam poplar, willow, and alder common in early succession floodplains. In most upland sites, conifers dominate in late succession, with birch and aspen in some upland sites. In general, white spruce occurs along flood plains and on well-drained upland sites, while black spruce occurs in sites with shallow soil, poor drainage, and/or shallow permafrost (Viereck et al. 1986).

Fire is the primary form of disturbance in Interior Alaskan forests, although disturbances from insects, wind, human harvest, and riparian erosion also occur. The distribution of dominant tree species has been shaped by past fire history. Fire is highly variable from year to year, depending on temperature and precipitation. Although the mean annual area burned is estimated at between 500,000 and 1 million ha, (Barney 1971; Dyrness et al. 1986), over 2.6 million hectares burned in 2004 followed by over 1.9 million ha in 2005.

The Kyoto Protocol and Carbon Sequestration Credits

In 1997, more than 160 nations drafted the Kyoto Protocol on Climate Change, a binding agreement under which developed nations agreed to limit their net release of greenhouse gases relative to 1990 levels (UN 1997). Its goal was to stabilize atmospheric concentrations of greenhouse gases to reduce the effects of anthropogenic climate change (IGBP 1998). Russia's ratification of the Kyoto Protocol in 2005 put the agreement into effect.

The Kyoto Protocol allows nations to use verified increases in carbon uptake and storage as credits against carbon release (Wilman and Mahendrarajah 2002; IGBP 1998). These credits are subject to complex accounting rules of net carbon stock changes and emissions of non-CO_2 greenhouse gasses, and are subject to country-specific caps. Nevertheless, signatory nations are allowed to trade or purchase carbon credits with other signatory nations (UN 1997). In January 2005, the European Union – including all 25 of its member states -- initiated a legally binding international trading market in greenhouse gas emissions (Kirk 2004).

Even though the US did not sign the Kyoto Protocol, carbon budgeting may become economically and politically important in the near future. The Chicago Climate Exchange (CCX) acts as a self-regulating voluntary market, administering the world's first multi-sector and multi-national emissions-trading platform. Trading volumes have topped 3,000,000 metric tons per month, and trading prices, although somewhat volatile, have generally risen since the Kyoto Protocol took effect, from about $1/tonne of CO_2 in 2004 to almost $4/tonne in May 2006) (CCX 2006). By participating in trading through CCX, corporations, municipalities, and other institutions have made legally binding commitments to reduce net emissions of greenhouse gases. Carbon emitters as well as credit holders are banking on the idea that the price of credits will escalate eventually, due to either international agreements or state and local laws. Some states, cities, and geographic regions are already making local commitments to reduce greenhouse emissions.

Goals and Objectives

In this study, my goal was to address the following questions:

1. What is the response of carbon storage in Interior Alaska forests to changes in fire cycles associated with scenarios of 0°C, 1.5°C and 4°C warming over 100 years?
2. How sensitive are these responses to assumptions regarding initial stand-age distribution?
3. How sensitive are these responses to alternative scenarios of temperature-induced changes in growth?

Further objectives of the study were to assess potential sources of uncertainty and error, and to examine how model outputs might impact forest and carbon management.

Methods

Overview

Two categories of models are currently available for estimating terrestrial carbon stores across wide areas: atmospheric models (e.g. Stocks et al. 1998, Prentice et al. 2000), Dargaville et al. 2002) and inventory-based or process-based models (e.g., Peng and Apps 1999; Kurz and Apps 1999; Kurz et al. 2002). In some cases a combined approach is used (e.g. Dargaville et al. 2002). In this study, I combine inventory-based and process-based approaches, integrating two models: the Carbon Budget Model of the Canadian Forest Sector (CBM-CFS3) and Alaskan Frame-based Ecosystem Code (ALFRESCO).

CBM-CFS3 is an inventory-based landscape-level model of forest carbon dynamics. It is intended to assess past or future stock changes in carbon based on forest parameters, which include age classes, dominant species, inventory, and growth-curves, that are coupled with specified disturbance regimes and management actions. The model was developed by the Carbon Accounting Team of the Canadian Forest Service (Kurz et al 1992; Kurz et al. 1996; Kurz and Apps 1999; Li et al., 2003) and is the core model in Canada's NFCMARS (Kull 2005). It is available for free use.

CBM-CFS3 simulates and tracks carbon pools by annual time steps in living biomass (including aboveground and belowground components, each divided into several pools) and dead organic matter (including snags, litter, the organic soil layer, mineral soil, and peat, with sub-pools within most of these categories). It accounts for the effects of growth, decay, and disturbance, including carbon released during fire or removed by harvest. Stand types are classified by species, age, site class and other classifiers selected by the user. The model is implemented at user-defined spatial scales from stand-level to the national scale. Ecological parameter sets are typically defined at the regional scale and do not explicitly consider spatial variability in climate within the region.

CBM-CFS3 was designed in the context of managed forestry, in a nation which is bound by the Kyoto Protocol to provide annual estimates of carbon balances required under the Kyoto Protocol (Kurz and Apps 2006). As such, the model's emphasis is on practical carbon accounting to guide forest management and planning within a context in which carbon sequestration is one of the economic, social, political and ecological goals.

CBM-CFS3 does not directly account for changes in forest dynamics and carbon fluxes attributable to ongoing climate change. Nor does it simulate the effects of ongoing climate change on disturbance (Kull 2005). However, the model provides the user with enough flexibility in assigning input parameters and adjusting model assumptions to allow for manual examination of alterations in these effects.

ALFRESCO is a frame-based spatially explicit model that simulates fire number and size in Interior Alaskan landscapes (Rupp et al 2000, 2001, 2002, in press). The model explicitly simulates ignition, fire spread, regeneration, and growth of dominant landscape cover types, using probabilistic rules. The physical characteristics of the landscape and the rules governing fire behavior and forest growth are based on data regarding species distribution, natural succession, fire spread, fire ignitions, temperature, and moisture.

In the version of ALFRESCO used for this research, the simulated landscape was a 300 km by 300 km grid at 1 km^2 resolution, based on ground-truthed data from the Ruby/Nowitna region in the heart of Interior Alaska (Figure 2.1). Of the total area of

9,000,000 ha, approximately 7,300,000 ha was forested. This region is geographically and climatically representative of the study region, with a typical range of cover types, stand ages, elevation, aspect, and drainage.

The model operated on an annual time-step. After fire, all forested cells (including deciduous, black spruce, and white spruce cover types) regenerated as early-succession deciduous forest. If such sites did not burn again during early succession, hardwood stands were succeeded by dominant conifers – either black spruce or white spruce – depending on site aspect. Succession to black spruce forest took place between ages 40 and 60, while succession to white spruce took place between ages 90 and 110. The fire regime of tundra and non-forested areas were simulated by ALFRESCO, but carbon balance was estimated only for forested cells.

Three separate simulations were performed in ALFRESCO using the same initial conditions. Model parameters were altered between simulations to mimic current climate conditions; modest warming (+1.5°C and +15% precipitation over 100 years); and more extreme warming (+4°C and +25% precipitation over 100 years). In all three simulations, it was assumed that humans were present on the landscape, affecting the number of fire ignition events. Within each simulation, temperature and moisture for each time step were determined stochastically and were based on statistical manipulation of past climate data, although in the two warming scenarios linear ramps in temperature and moisture were superimposed on this interannual variability. Thus, fire patterns varied widely from year to year, mimicking the natural irregularity seen in the region. In order to maintain this interannual variability within the context of 100-year trends, ALFRESCO outputs used as CBM-CFS3 inputs were the result of single simulations, rather than composites of multiple runs.

Modeling Interior Alaska

In order to create climate-dependent landscape-level simulations, I first assessed the parameters used within CBM-CFS3 in order to determine which would require modification in order to accurately represent Interior Alaska. I adjusted these parameters according to the best available Alaska-specific data in the published literature (e.g. Yarie and Billings 2002). I then created model parameters and formulas based on published data to represent forest conditions, inventory, and growth under present and projected climate scenarios, and formatted these inputs appropriately for CBM-CFS3. I used ALFRESCO simulations to create input data for fire dynamics under all three climate scenarios. I then used all the above input data to perform CBM-CFS3 simulations for each climate scenario and analyzed and compared the results from the three different sets of climate assumptions. Finally, I re-ran the model under a range of different parameters in order to examine the sensitivity of model responses to (1) different initial stand-age distributions and (2) different scenarios of growth responses to climate change.

The ecosystem type already parameterized within CBM-CFS3 and most similar to Interior Alaska is the Boreal Cordillera of the Yukon Territory in Canada (Figure 2.2), which is contiguous with Interior Alaska and has all of the most abundant tree species that occur in Interior Alaska. Temperature, moisture, soil conditions, permafrost, and growing season are also similar. Previous efforts to systematize the boreal forests of North America have placed these regions within a single environmentally determined stratum (Botkin and Simpson 1990). In using CBM-CFS3 to model carbon dynamics in Interior Alaska, I used Yukon Cordillera parameters as a starting point. However, whenever possible, I altered model parameters based on studies conducted in Interior Alaska.

Model inputs necessary in the CBM-CFS3 framework include forest inventory by species (stand type) and area; volume/age curves for each stand type; disturbance by stand type and age, area, and year; and forest transition rules post-disturbance. Parameters already included in the model include rates of litterfall and decomposition of all components of biomass and soils; volume to biomass conversion; and transformations of biomass and soils resulting from disturbance. These internal parameters are based on extensive fieldwork and model verification (Kurz and Apps 1999).

The parameters used by CBM-CFS3 to represent carbon dynamics in the Yukon Cordillera are based on estimates of prevailing climate variables for that region. Thus, I first ascertained that climate variables in Interior Alaska are comparable. I analyzed the climate in Interior Alaska using data from the Western Regional Climate Center (WRCC 2005). I selected five communities representing the range of geographic and climatic variation across Interior Alaska (Figure 2.1) and compared all available historical mean temperature and moisture data from these communities with the mean values assigned by CBM-CFS3 for the Yukon Cordillera (Figure 2.3). The mean annual temperature across all five communities (-4.43°C) is almost identical to the mean temperature used as a model parameter for the Yukon Cordillera by CBM-CFS3 (-4.41°C). Thus, I used the default mean annual temperature for my "no climate change" model runs, and adjusted this to -2.91°C for the "+1.5°C" runs and to -0.41°C for the "+4°C" runs.

Precipitation data show greater divergence between Interior Alaska and the Yukon Boreal Cordillera. However, the value used by CBM-CFS3 (37.796 cm) is within one standard deviation of the Interior Alaska mean calculated from the five selected communities (29.88 cm). The currently available version of the model does not link mean annual precipitation to forest decomposition or growth. Thus, I left this parameter unchanged in all model runs.

I also used Yukon parameters for mean annual insolation and mean length of growing season, even though the Yukon Cordillera is at slightly lower average latitude than Interior Alaska, because between-region differences are small relative to within-region differences.

CBM-CFS3 requires user-input inventory data and growth curves for all stand types. Stands may have a single dominant species, or may be composed of several species, and the early-successional hardwood component of spruce stands can be explicitly modeled. Understory tree species are included in underlying model parameters and did not require separate parameterization.

In order to parameterize forest inventory and growth, I chose to use published values separate from the inventories used in the ALFRESCO simulations in order to include the broadest possible range of data from the entire study area. Reliable data on both stand age distribution and species-specific growth in Interior Alaska are extremely limited. Thus, I relied on landscape-level averages from Interior Alaska and later tested the model for sensitivity to uncertainty in these parameters. I used published US Forest Service data on timber resource statistics from Interior Alaska (van Hees 1984; 1987; Hegg 1975; Setzer 1987), as well as a comprehensive compilation and analysis of many of these data (Yarie and Billings 2002); biomass equations used in the compilation of the data (Yarie et al. unpub.); and normal yield tables for boreal species (Plonski 1956). Forest Service inventories were performed over the past 50 years and thus represent average values rather than values specific to a particular date. The surveys covered a broad range of spatial variability in the Interior, including the Fairbanks and Tanana area (central Interior); the Porcupine and Upper Yukon regions (northeastern Interior); the Copper River and Upper Tanana areas (southeastern Interior) and Western Alaska.

It was necessary to reconcile inventory and growth data with model inputs for fire disturbance events derived from ALFRESCO. In the ALFRESCO model, all deciduous stands transitioned to white spruce (at ages 90-110) or black spruce (at ages 40-60) if not disturbed in early succession, in a ratio of approximately 1.6 to 1. In contrast, growth and inventory data classified birch and other hardwoods as discrete categories, with some stands older than 110 years (Yarie and Billings 2002; van Hees 1984; 1987; Hegg 1975; Setzer 1987).

In order to simplify model inputs while maintaining a close approximation of real-world conditions, I parameterized CBM-CFS3 inputs for black spruce, white spruce, and Alaska birch, the three most prevalent tree species in Interior Alaska. All of these species were available as preexisting types in the model's Yukon Boreal Cordillera spatial unit. I identified birch stands from the inventory data as either early successional black spruce or early successional white spruce at a ratio of 1 to 1 prior to age 60, and as early successional white spruce only between age 60 and 110. I included no inventory data for birch stands older than 110, since such stands were not part of the landscape structure simulated by ALFRESCO.

I divided each stand type into twenty-year age classes up to age 200. Due to the frequency of fire on the landscape, stands over 200 years old are relatively rare. Inventory data for stands older than 200 were included in the oldest age class.

Yarie and Billings (2002) calculated the total area of each stand type and age in Interior Alaska, based on extrapolation of existing inventory data. I created second-degree polynomial regressions to show the pattern across age classes for each species (Figure 2.4). I then used these regressions to calculate initial stand areas by age class for baseline model runs. The area dominated by birch drops off steadily after approximately age 60 despite low flammability in deciduous stands, supporting the model assumption that birch succeeds to black spruce or white spruce. For the same reason, there are few young stands of black or white spruce. Cumulative loss to fire in black spruce and white spruce stands accounts for the relative lack of older coniferous stands.

Lack of young age classes (<40 years) of all species in the data may reflect true landscape conditions or a flaw in the data (either bias in stand selection or a mis-classification of young stands as non-forested). The latter hypothesis is suggested by the more biologically reasonable decreasing patterns across age-classes associated with higher R^2 values, when the two youngest age classes are omitted (Figure 2.5). I used these regressions to generate initial conditions for model sensitivity simulations. The pattern shown in Figure 2.5 may, however, overestimate total forest area and stands >150 years. The pattern shown in Figure 2.4 is, however, consistent with stand-age reconstructions from Interior Alaska, showing that, for unknown reasons, stand ages have gradually increased during the 20th century, leading to a low frequency of young stands on the landscape (Duffy 2006). Fire suppression is unlikely to explain the low frequency of early-successional stands because it has never been universally applied at the landscape level in Interior Alaska. A combination of the above effects is likely, i.e. a reduced number of young stands on the landscape coupled with undercounting of those present.

Overall variability among age classes in the published data may be due to variability in fire cycles. Large fire years tend to lead to a plethora of stand reinitiation in the year or two immediately following. Analysis of model sensitivity to uncertainty in inventory data is described below.

The total area of boreal forest in Interior Alaska has been variously estimated as 17,244,098 ha (Yarie and Billings 2002); between 9,000,000 and 43,000,000 ha (Hutchinson 1968) and between 9,000,000 and 53,000,000 ha (Burton et al. 2003), depending on discrepancies in definitions of "productive forest" and differences in ecoregion boundary definitions. For the purposes of this paper, I relied on lower-end estimates within this range, since I wanted to exclude tundra, rock, ice, treeless bog, grassland and other non-forested land within the ecoregion. The total area classified as black spruce, white spruce, or birch (the majority of the area of productive hardwoods recorded) and identified by age class by Yarie and Billings (2002) covered 11,913,325 ha. I used this area as a reasonable estimate of productive forest land in Interior Alaska.

In order to calculate growth curves for black spruce and white spruce stand types (including their respective deciduous early-succession components), I grouped the available age-specific volume data into the above age classes, and then fit a polynomial regression curve to each species similar to growth curves used to generate normal yield tables (Plonski 1956, Yarie and Billings 2002). I used a second-order equation to simulate these species' pattern of relatively rapid and accelerating early live biomass accumulation that slows and even becomes negative as early-succession deciduous species and then older spruce senesce and move to the dead biomass or litter pools (although total ecosystem carbon nonetheless continues to increase in these older stands) (Figure 2.6).

Because CBM-CFS3 is a timber-based model, it requires biomass input in the form of merchantable volume per hectare, in cubic meters. Since available data were expressed as gross biomass rather than merchantable volume, I converted weights to volumes using standard density tables for boreal species (Runesson 2002). I then shifted each species curve according to ratios derived from the relationship between total volume and merchantable volume curves in Normal Yield Tables (Plonski 1956). Across site classes in the boreal zone, the relationship between merchantable volume and total volume for spruce and for birch are shown in Figure 2.7.

I formulated growth curves for the deciduous and coniferous components of each stand type by using the yield estimates described above, assuming birch dominance for 40-60 years in black spruce stands and 90-110 years in white spruce stands (Figure 2.8). I used these growth curves as a baseline for all of the adjusted growth curves that I tested for sensitivity of growth responses to climate change, as described below.

I simulated disturbance events based on ALFRESCO model output. Both ALFRSCO and CBM-CFS3 operate on an annual time step. However, ALFRESCO is spatially explicit while CBM-CFS3 is not. I used a simple R-script to convert map data outputs into tables of annual area burned by age class and cover type for each year of the 100-year model runs. I then input these data into CBM-CFS3 such that each combination of year, age class, and area was counted as a separate disturbance. In keeping with the assumptions of ALFRESCO, all fires were considered stand-replacing, and dominant forest cover type did not change after fire. Thus, all stands returned to age 0 after fire, and subsequent early-succession hardwood growth commenced with no delay, but stands retained their late-succession classification as either white spruce or black spruce. ALFRESCO runs took place on a 9,000,000 ha simulated landscape, but I was using an area of 11,913,325 ha to approximate the total productive forest land in Interior Alaska. Thus, I scaled up all disturbance areas by a factor of 1.324.

In baseline model runs, neither decay rates nor forest growth rates were directly affected by climate change; differences between runs were based solely on differences in fire disturbance. However, the CBM-CFS3 currently uses a combination of Q10 and decay rate parameters to simulate the dynamics of the above- and below-ground slow pools. The values were derived through statistical analysis to fit a national-scale dataset of soil C values (Kurz, pers. comm.). The best statistical fit was achieved for the slow soil C pool with a Q10 of 0.9 and a decay rate of 0.0032 (at 10 °C reference temperature). For simulations involving a single geographic region and climate change scenarios, using a Q10 of 2 is more appropriate. I changed the Q10 to 2 (similar to all other dead organic matter pools in the model) and set the decay rate to 0.00372 at -4.41 °C reference temperature. Thus, the decay rates at -4.41°C remained unaltered while allowing for a doubling of decay rates for all carbon pools with every 10°C increase in temperature.

Sensitivity Analysis

In order to address the sensitivity of model output to uncertainties regarding initial forest age-class structure and alternative assumptions regarding the growth response of forest biomass to climate warming, I performed additional model runs using alternative scenarios for initial forest inventory and species growth curves. I then compared the results of these simulations with those of the primary model runs described above.

I selected two alternative scenarios for initial stand inventory. Each was based on the same total land area (11,913,325 ha, based on Forest Service data) and the same relative abundance of white spruce and black spruce stands (1:1.0036) used in the primary simulations. In the first alternative scenario, area was distributed uniformly across all stand age-classes. In the second case, in order to correct for potential bias in undercounting young forest stands, area was distributed according to the functions shown in Figure 2.5, in which second-order polynomial regressions were fit to all inventory classes above age 40. I performed model runs with no climate change using each of these alternative scenarios, and compared the results to the baseline scenario.

To test the sensitivity of the model to assumptions regarding the response of growth and yield to changing climate, I performed model runs using three new sets of growth curves. In the first case, growth of all species was assumed to increase by 5% above the baseline for each 1°C increase in temperature (for a total increase of approximately 20% in the +4°C scenario). This magnitude of response is consistent with that suggested by some previous simulations (Keyser et al. 2000). Growth curves were adjusted annually, based on the mean annual temperatures used in the ALFRESCO simulations; thus, for some years temperatures and growth curves dropped below the baseline. In order to simulate a more extreme response, I also performed model runs in which growth curves were increased by 10% per degree temperature increase. Finally, in order to simulate a scenario in which white spruce on moisture-limited warmer upland sites suffer from climate change while black spruce in colder wetter habitats increase NPP (Barber et al. 2000; Juday et al. 2003), I created growth curve sets in which black spruce growth increased by 5% per degree while white spruce growth decreased by 5% per degree.

RESULTS

Baseline Simulations

Model runs simulating the impacts on ecosystem carbon dynamics of the three different climate-related fire scenarios showed important differences in their results. In the simulations with no climate change or mild climate change, total ecosystem carbon stocks increased about 4% from approximately 1.91 Gt to approximately 1.98 Gt over the 100-year model run, while in the simulation with a four-degree increase in mean annual temperatures, total ecosystem carbon stayed roughly the same, increasing by only 0.5% over same time period (Figure 2.9). Although the modest increase in carbon storage in the no-climate-change scenario may reflect the model's sensitivity to assumptions regarding starting inventories, as discussed below, the differences between the three climate scenarios can be directly attributed to changes in climate-induced fire. A total of 824 million ha burned over 100 years in the +4°C scenario, whereas only 645 million ha burned over the same time period in the +1.5°C scenario and 610 million ha burned in the no-climate-change scenario. The lack of significant difference between the area burned in the latter two scenarios reflects in part the stochastic nature of fire starts and fire spread which were simulated independently for each scenario by the ALFRESCO model.

Due to relatively similar patterns of carbon accumulation in soils under all climate scenarios, differences in overall carbon dynamics could be attributed largely to changes in plant biomass and fast-decaying components of the DOM pool (Figure 2.10). Biomass carbon and DOM carbon together account for all ecosystem carbon. Soil carbon, which is a subset of DOM carbon and contains pools with slower decay rates and less susceptibility to fire, increased in all scenarios. Increased incidence of fire under the

+4°C fire regime led to decreases in biomass across the 100-year simulation At the end of the simulation period, more young stands (<40 years) and fewer older stands (>180 years) were present for this scenario, as compared to the scenarios with no warming or moderate warming (Figure 2.11).

Much of the biomass change across the 100-year simulation period took place during high fire years (Figure 2.12). These occurred sporadically, according to the stochastic nature of inputs derived from the ALFRESCO model. However, more extreme climate change was generally associated with greater size and frequency of fire. Total change in ecosystem carbon was negative (carbon loss from the system to the atmosphere) in 19 of the 100 years in the no-climate-change scenario, in 20 years with +1.5°C change, and in 24 years with +4°C change.

Sensitivity to Assumptions Regarding Initial Age-Class Structure

Changes in the age-class structure of initial forest inventory had relatively little effect on overall model outcomes in terms of total carbon storage at year 100 of the simulations, but altered how strong a carbon sink the system was during the model runs (Figure 2.13). Starting inventories skewed towards younger age classes (inventory from fitted curve; Fig. 5) showed the greatest increases in carbon stocks, while starting inventories skewed towards older age classes (even inventory) showed lesser increases. However, carbon storage in all three model runs tended to converge as age-class structure converged (Figure 2.14). Variations in total ecosystem carbon stocks attributable to alterations in initial inventory inputs were most evident early in model runs, but by year 100, forest age structure, and thus total carbon, was determined more by fire disturbance than by initial inventory. In model runs that started with a young age-class distribution and shifted to an older distribution, total ecosystem carbon increased by about 6%, while in the model run in which initial stand ages were evenly distributed, total ecosystem carbon increased by only 2%, despite the relatively low fire frequency associated with the no climate change scenario.

Sensitivity to Scenarios of Growth Responses to Climate Change

Not surprisingly, CBM-CFS3 outputs proved to be sensitive to differing assumptions regarding temperature-induced changes in forest growth and yield (Figure 2.15), since such alterations directly shifted total forest biomass and indirectly shifted all other carbon stocks. Growth assumptions played a clear role in carbon accumulation in both the climate-change simulations, although the magnitude of change was relatively small in the less extreme warming scenario. In all cases, long-term shifts attributable to growth assumptions were superimposed upon annual shifts attributable to fire disturbance. In the +4°C scenario, when growth responses to climate change were mixed (i.e. an increase of 5% per °C in black spruce, but an equivalent decrease in white spruce), total ecosystem carbon at the end of the simulation was essentially the same as that at the beginning. However, when a growth response of +10% per °C was assumed, total ecosystem carbon increased by approximately 3%.

At the end of 100-year model runs, biomass carbon remained relatively unchanged in the no climate change scenario, and increased with slight warming, particularly when the growth response to warming was strongly positive (Table 2.1). With more pronounced warming, increases due to growth response were not great enough to offset losses due to increased fire. Carbon in dead organic matter and total ecosystem carbon increased for all scenarios.

DISCUSSION

This study sought to quantify the response of carbon storage in Interior Alaska forests to changes in fire cycles associated with climate change; to assess how sensitive these responses were to assumptions regarding initial stand-age distribution and alternative scenarios of temperature-induced changes in growth; to analyze potential sources of uncertainty and error; and to examine how model outputs might impact forest. Due to uncertainties in the forest inventory data required to characterize the initial forest conditions, more robust conclusions can be drawn regarding the differences between climate scenarios and the general magnitude of sequestration across all scenarios than regarding the exact sequestration in any one scenario. However, clear patterns with potential policy implications did emerge.

Model results showed the forested landscape of Interior Alaska acting as a weak carbon sink in simulations with no climate change and those with +1.5°C climate change, and as a very weak sink for atmospheric carbon under simulations of more severe climate change. Using the best available data as nominal inputs for forest growth and inventory, model outputs showed a net sequestration of approximately 68Mt of carbon in an area of 11,913,325 ha over 100 years (0.57g/m^2/yr) without climate change or with a 1.5°C temperature increase and accompanying change in fire cycles, and a net sequestration of 0.09g/m^2/yr with a 4°C temperature increase. Total biomass carbon for Interior Alaska, including aboveground and belowground components, ranged from 391-431 million Mg, depending on climate assumptions. These estimates are similar to the 476 million Mg estimated by Yarie and Billings (2002). The slightly lower values obtained in this analysis (9-18%) partially reflect the lower estimates (31%) of total forested land area in the Interior.

Based on model results, forested Interior Alaska currently appears to lie close to the boundary between being a net carbon sink and a net source, and the frequency of climate-triggered high-fire years has a strong impact on system carbon. Results also show that shifts towards an older age-class structure (as might occur through fire suppression) are associated with system-wide increases in carbon storage while the shift is occurring, while negative growth responses to climate change lead to comparative carbon losses.

In summary, increases in factors leading to system carbon loss might cause Interior Alaskan forests to cross the source/sink boundary and become a net source of atmospheric carbon. For example, climate change of greater than +4°C, negative growth response to climate change, or more pronounced exacerbation of fire cycles by warming might contribute to such an effect.

As previously noted, growth responses are particularly difficult to predict. Some modelers have predicted that NEP could increase by more than 25% as a result of climate change (Kasischke et al. 1995; Yarie and Billings 2002). Other research shows decreased growth in productive white spruce stands due to drought stress (Barber et al. 2000; Juday 2003) or climate-induced pest outbreaks (Juday 1998; Volney and Fleming 2000). In contrast, Yarie and Parton (2005) found a more complex response to warming in which white spruce showed increased carbon capture with warming, hardwoods showed decreased carbon capture, and black spruce showed an age-dependent response.

In the case of continuously increasing fire disturbance, one might expect to see ongoing loss of carbon from the system. However, if fire cycles were to stabilize again at some higher average frequency, the predicted result would be stabilization of ecosystem carbon sequestration, albeit at smaller quantities than current levels. Likewise, reducing the mean frequency and size of fires in the Interior through active suppression would be expected to raise total ecosystem carbon levels to some new stable point.

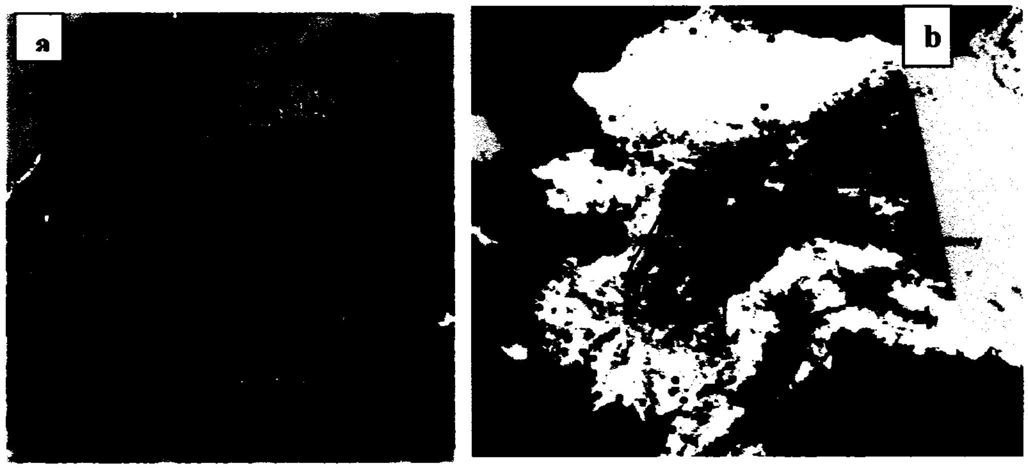
The results obtained from CBM-CFS3 might underestimate the potential effects of climate change on terrestrial carbon loss in several ways. First, although mean temperatures remained below freezing in all scenarios, landscape variability is high in the Interior, and even minimal warming is predicted to result in significant loss of permafrost (Jorgenson et al. 2001, Hom 2003, Hinzman et al. 2006). This, in turn, is likely to lead to lowered productivity on some newly thawed areas, due to subsidence and soil saturation, as well as more rapid loss of slow pool of soil carbon through decomposition in areas where soils are not saturated. Second, although CBM-CFS3 accounts for changes in decomposition associated with changes in temperature, it does not account for changes in decomposition associated with changes in soil moisture associated with permafrost thaw and changes in precipitation.

On the other hand, since non-forested landscapes were excluded from this analysis, model results do not include potential increases in carbon sequestration potential in area in which tree line is moving north, or tundra areas within current forest boundaries are being filled in by forest cover. Treeline shift might be accelerated by positive feedbacks to localized warming due to changes in albedo; at the same time, the lower albedo of young hardwood stands in recently burned areas might provide a negative feedback to warming (Chapin et al. 2000).

From the standpoint of obtaining tradable carbon credits, landscape-level forest management appears to be a poor option due to the unreliability of predictions (due to exclusion of changes in hydrologic cycle and successional trajectory from the model), the relatively small magnitude of potential carbon gains, and the expense and risk of attempting to gain credits through a program of continuous fire suppression. Ongoing climate trends are pushing us towards a situation in which Alaska's boreal forests may shift from being an overall carbon sink to being a source. Similar changes may have already occurred in the Canadian boreal forest (Kurz & Apps 1999). Intensive fire suppression might curtail or reverse this trend, but carbon sequestered under such a management system would be eligible only for a one-time credit. Fire frequencies would then have to be perpetually maintained at the lower levels in order to maintain a higher level of carbon storage. The costs of fire suppression in Alaska are already high (Todd and Jewkes 2006) and are likely to become higher if population growth and new development make human-caused ignitions more common and if climate change increases the area annually burned.

Conclusion

In general, simulation of landscape-level carbon dynamics in Interior Alaska using ALFRESCO fire simulation data as disturbance inputs in CBM-CFS3 showed lowers rates of landscape-level carbon sequestration under warming scenarios. Model results showed slight net gains in terrestrial carbon under unchanging climate conditions or mild warming but ambiguous results that depended on assumptions of growth response to temperature under more significant warming conditions. The similarity of these results to those obtained for Interior Alaska and other boreal regions using other modeling approaches provides support for these conclusions. Since most current predictions forecast that climate change in the far north will be relatively severe, the +4°C warming scenario appears to be the most likely, and thus the most pertinent for land management planning purposes. The costs and risks associated with the level of fire suppression necessary to reverse the observed trends would likely far outweigh the benefits obtained from carbon credits. As such, under current regulatory and market conditions, landscape-level management of fire disturbance is unlikely to prove to be a reliable or advisable means of obtaining tradable carbon credits in Interior Alaska. Instead, managers might choose to explore smaller-scale projects such as obtaining fuel offsets through harvest and regrowth of biomass fuel (Chapter 3).



Brown = deciduous
Dark green = black spruce
Light green = white spruce
Tan = Tundra
White = no vegetation

Green = forested area

Area of interest demarcated by black line

Figure 2.1. ALFRESCO initial vegetation map (a) for simulations in the Ruby/Nowitna area in the heart of Interior Alaska (b) (adapted from Crimp and Adamian 2001).

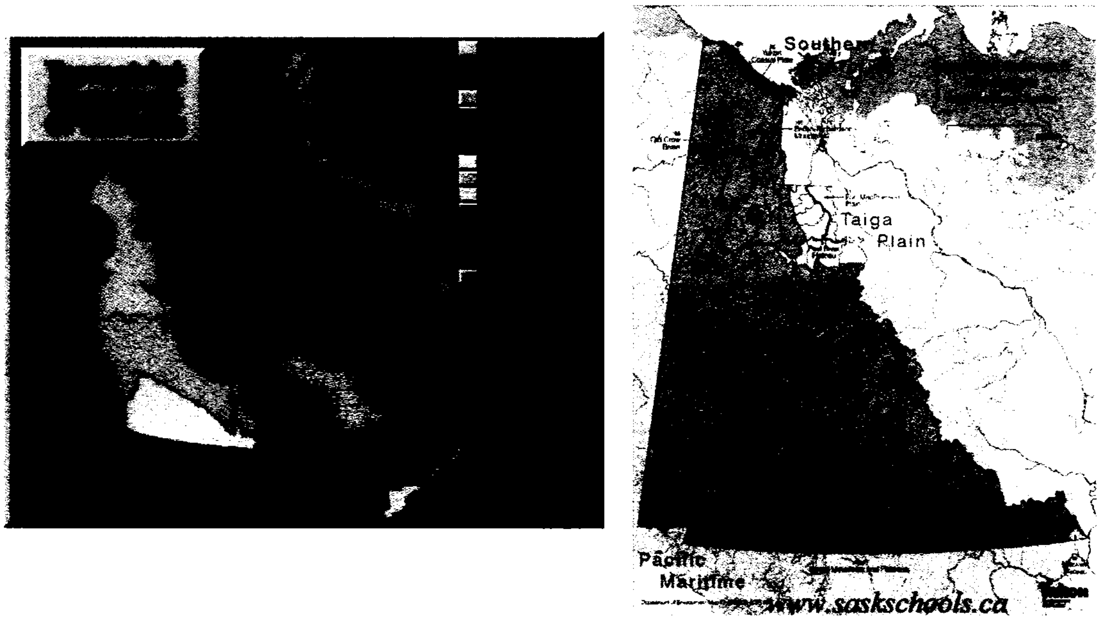


Figure 2.2. Ecozones of Canada and the Yukon Territories. The Boreal Cordillera of the Yukon Territories, shown in blue on both maps, is spatially contiguous with the Interior Alaska boreal forest.

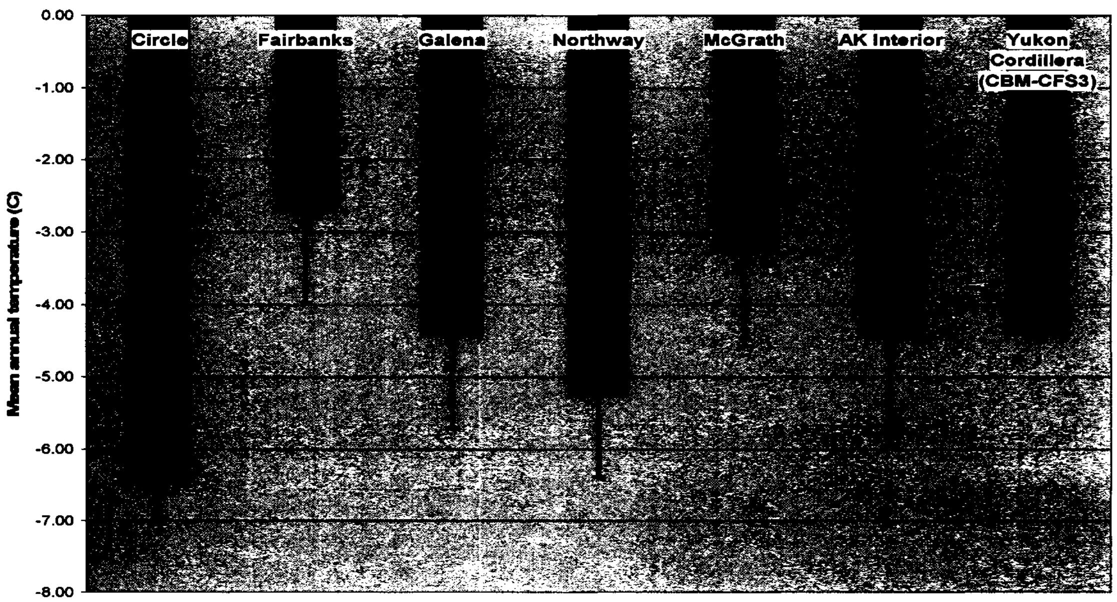


Figure 2.3. Mean annual temperature for selected communities in Interior Alaska. Values are based on available records from 1942 to the present (WRCC 2005).

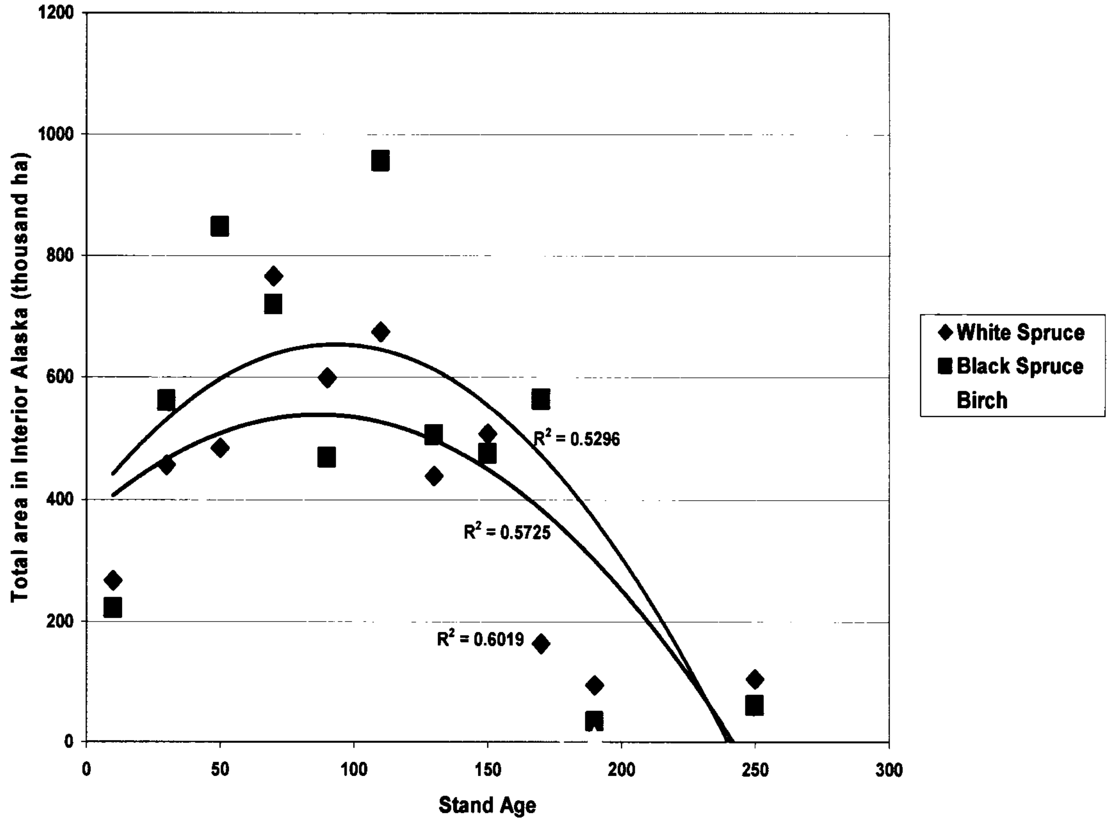


Figure 2.4. Total area by forest stand type and age for Interior Alaska.

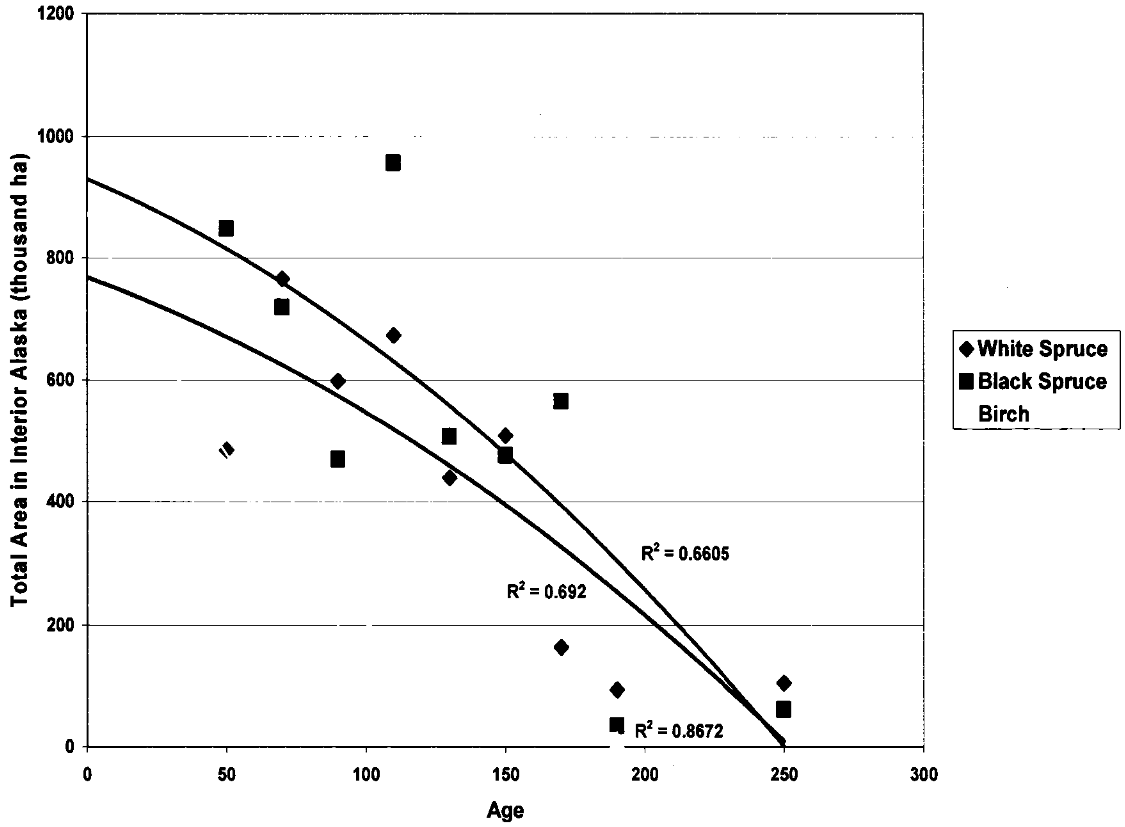


Figure 2.5. Total area by forest stand type and age omitting ages <40 years.

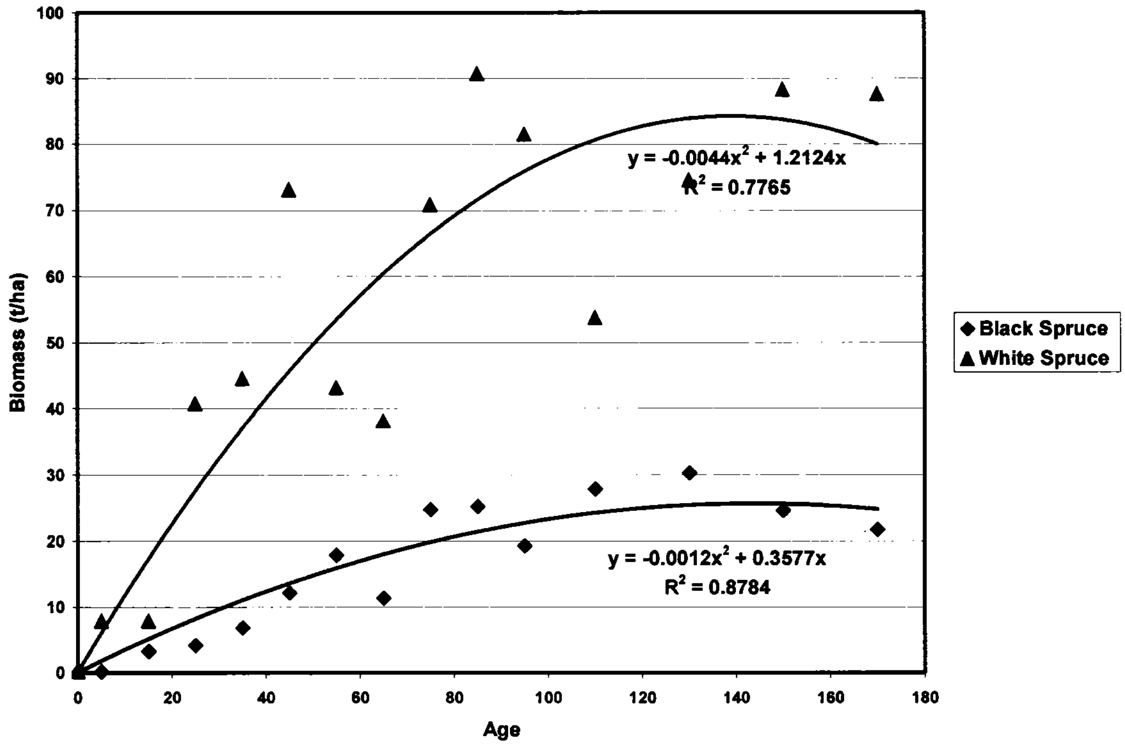


Figure 2.6. Aboveground biomass growth curves for the dominant stand types in Interior Alaska.

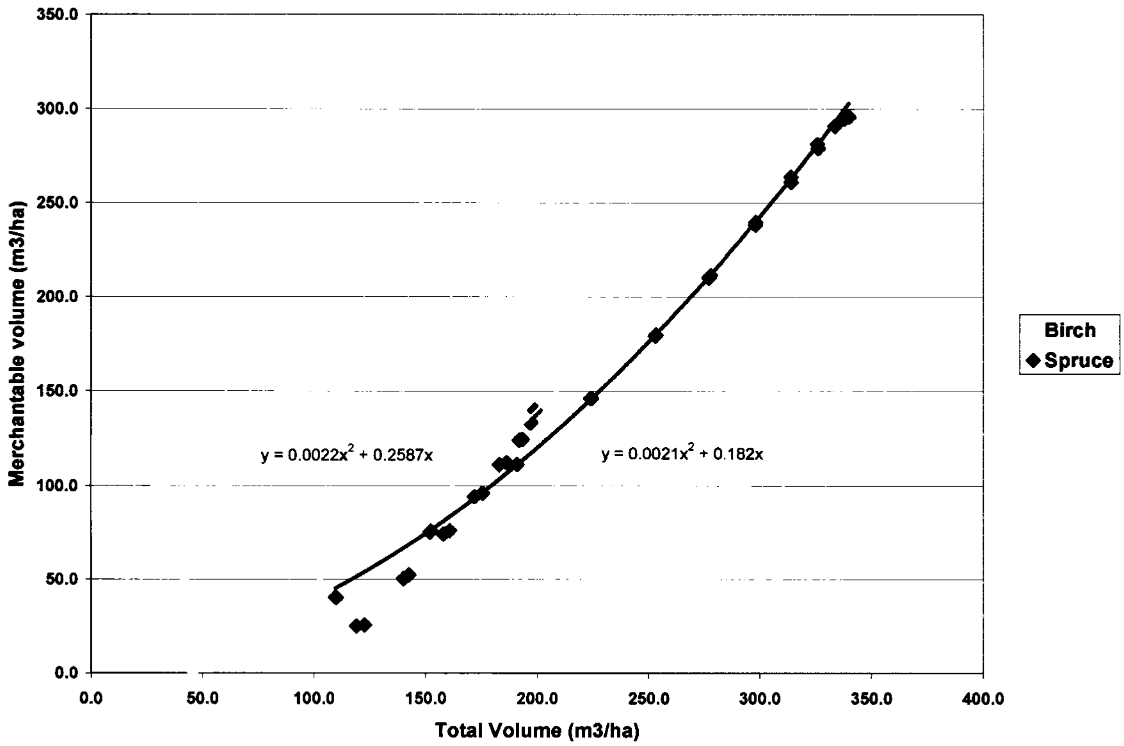


Figure 2.7. Relationships between total volume and merchantable volume for spruce and birch. These relationships are derived from data from a range of site classes and stand ages in the boreal forest of northern Ontario. Both black spruce and white spruce stands are included in the spruce data (Plonski 1956; Runesson 2002).

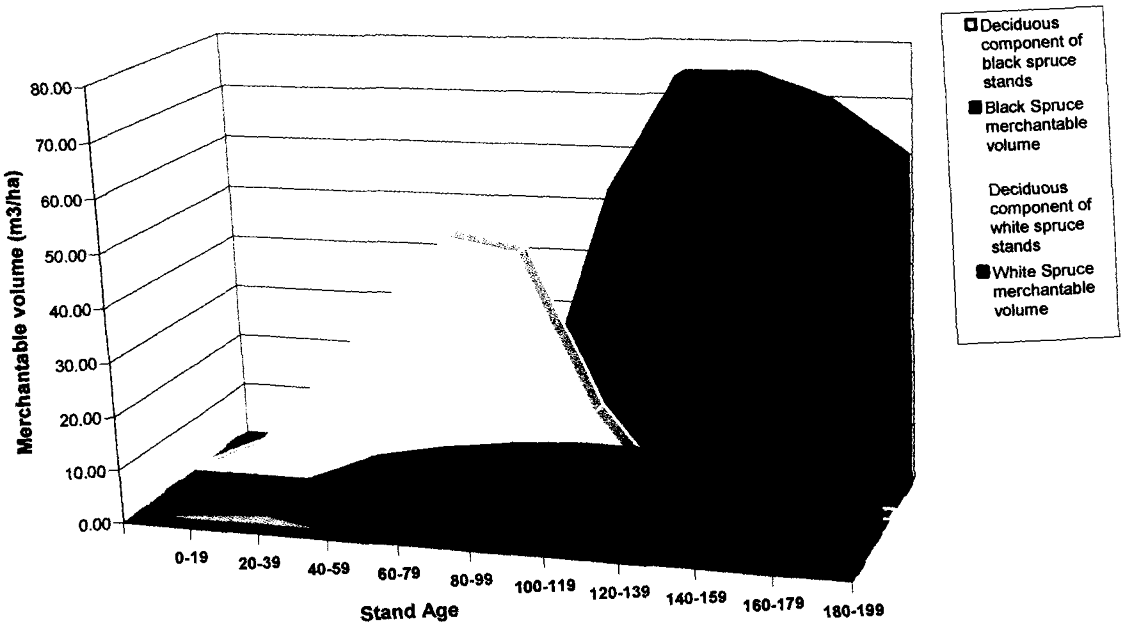


Figure 2.8. Growth curves for black spruce and white spruce stands. In each stand type, an early-succession deciduous component (Alaska birch) is replaced by the leading species in later succession.

Figure 2.9. Total ecosystem carbon under three fire/climate scenarios. Total carbon includes all living and dead biomass above and below ground, including all soil pools. Significant differences among all three model runs reflected differences in disturbance events.

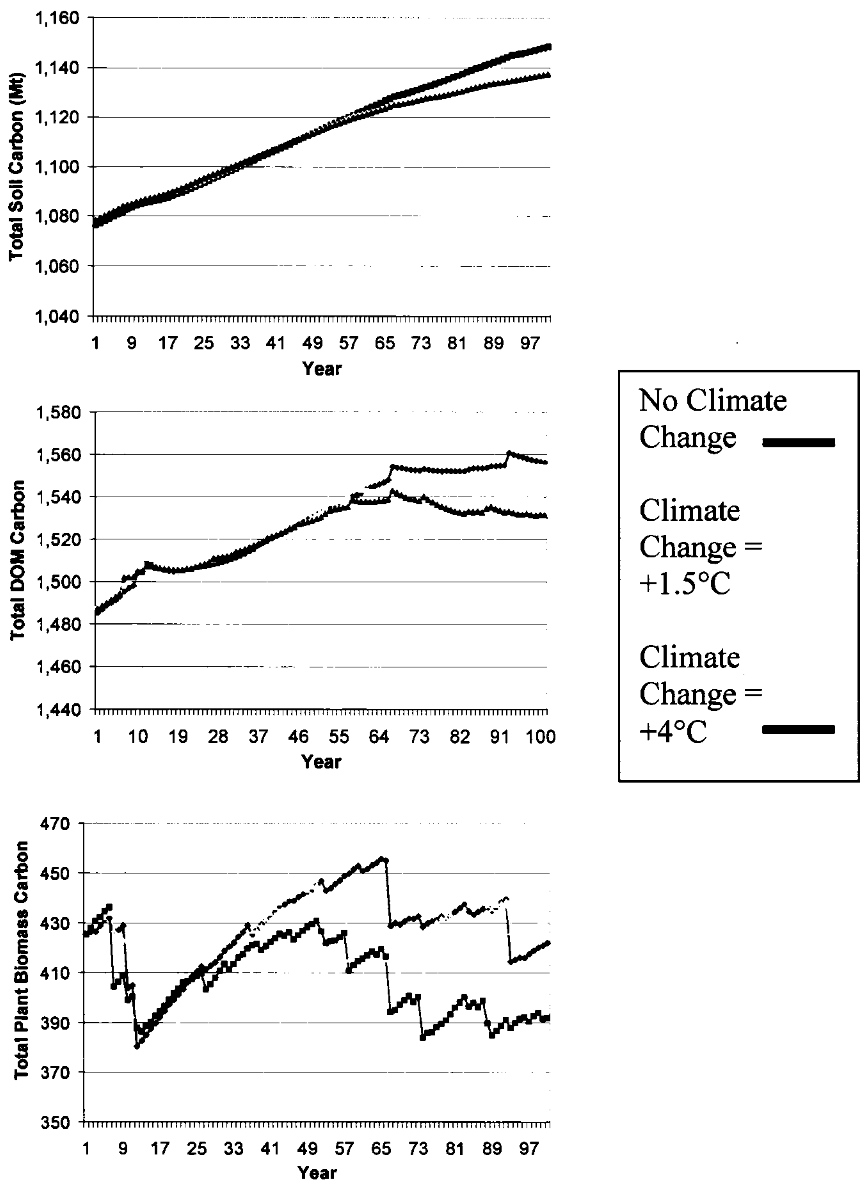


Figure 2.10. Total soil carbon, DOM carbon and biomass carbon under three climate/fire scenarios. DOM carbon increased early in the model run in all scenarios, but showed later decreases under more severe climate change despite continued increases in soil carbon, a subset of the DOM pool. Biomass carbon remained relatively unchanged under no climate change or moderate climate change, but dropped by 8% with 4°C warming.

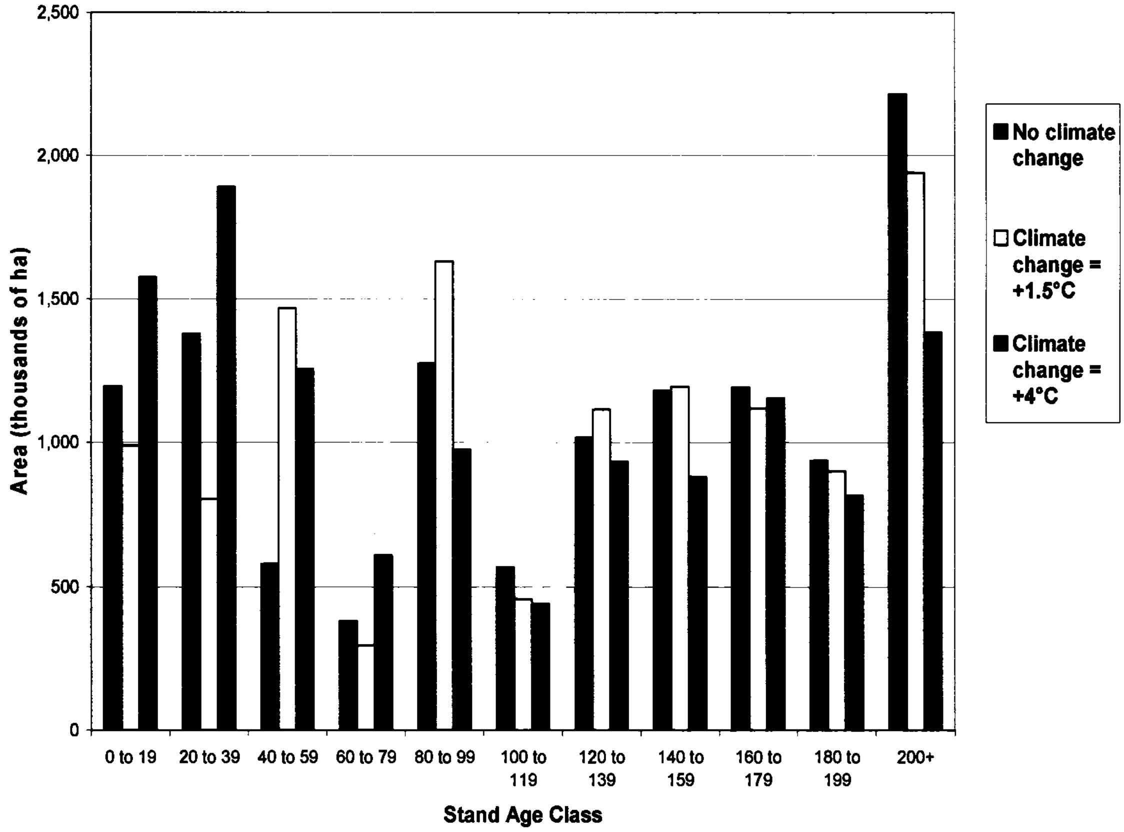


Figure 2.11. Forest area by stand age class at year 100 of simulations for three climate scenarios. More frequent fire associated with the +4°C scenario resulted in a higher proportion of younger stands, and fewer stands in the oldest age classes.

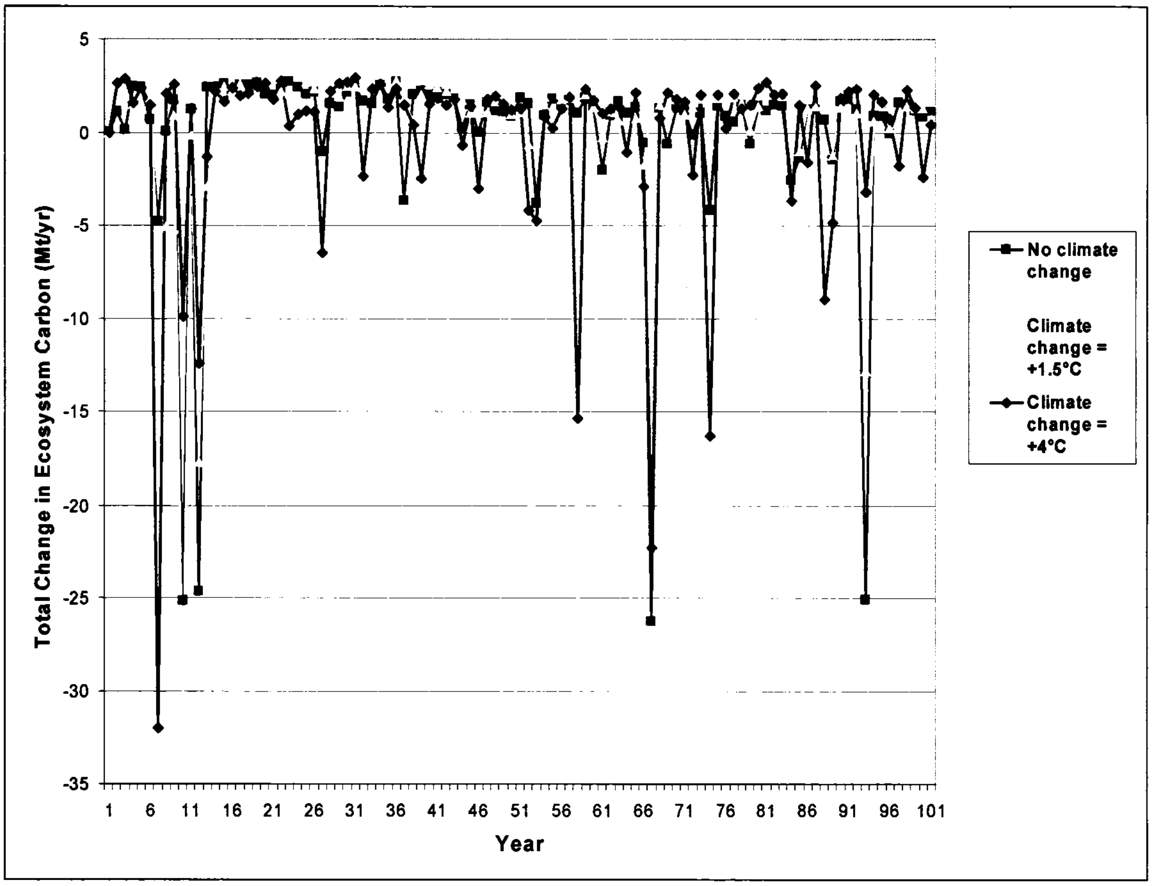


Figure 2.12. Annual change in ecosystem carbon under three fire/climate scenarios. Severe fire years account for the majority of carbon loss the system through disturbance.

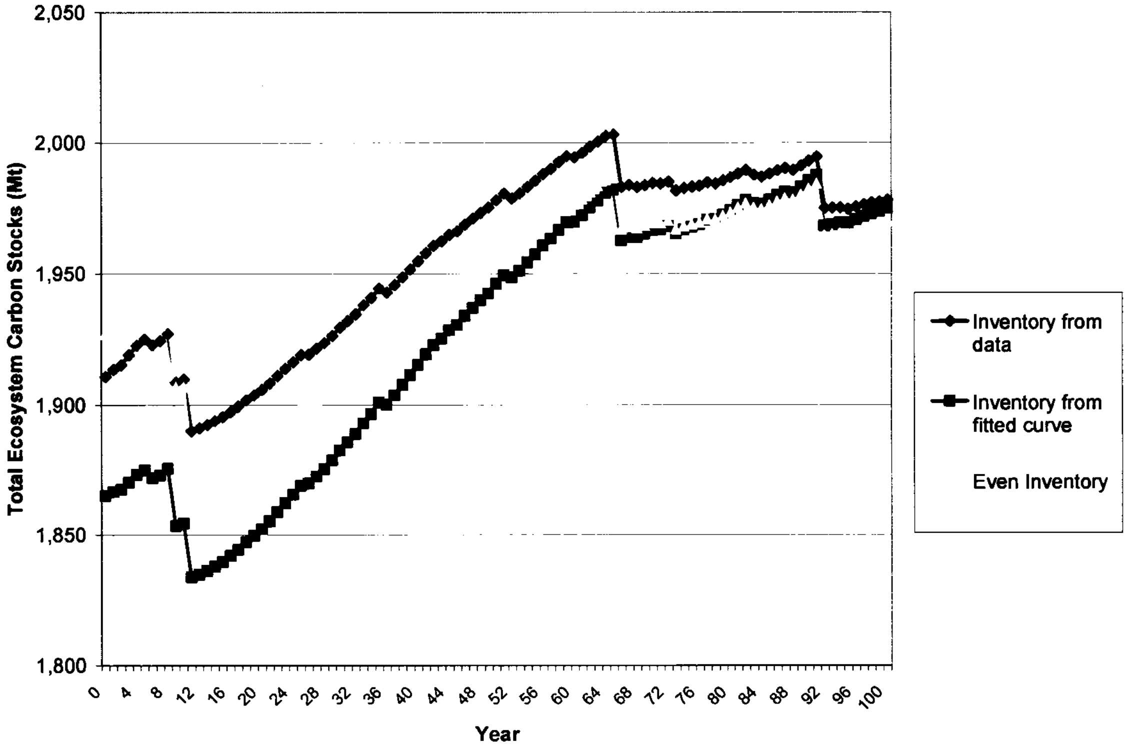


Figure 2.13. Sensitivity analysis of initial stand-age structure inputs. Each model run represents an iteration of the no-climate-change scenario, using different model inputs for forest inventory in year zero. Although starting inventory causes variation among model runs, by year 100 total ecosystem carbon has converged.

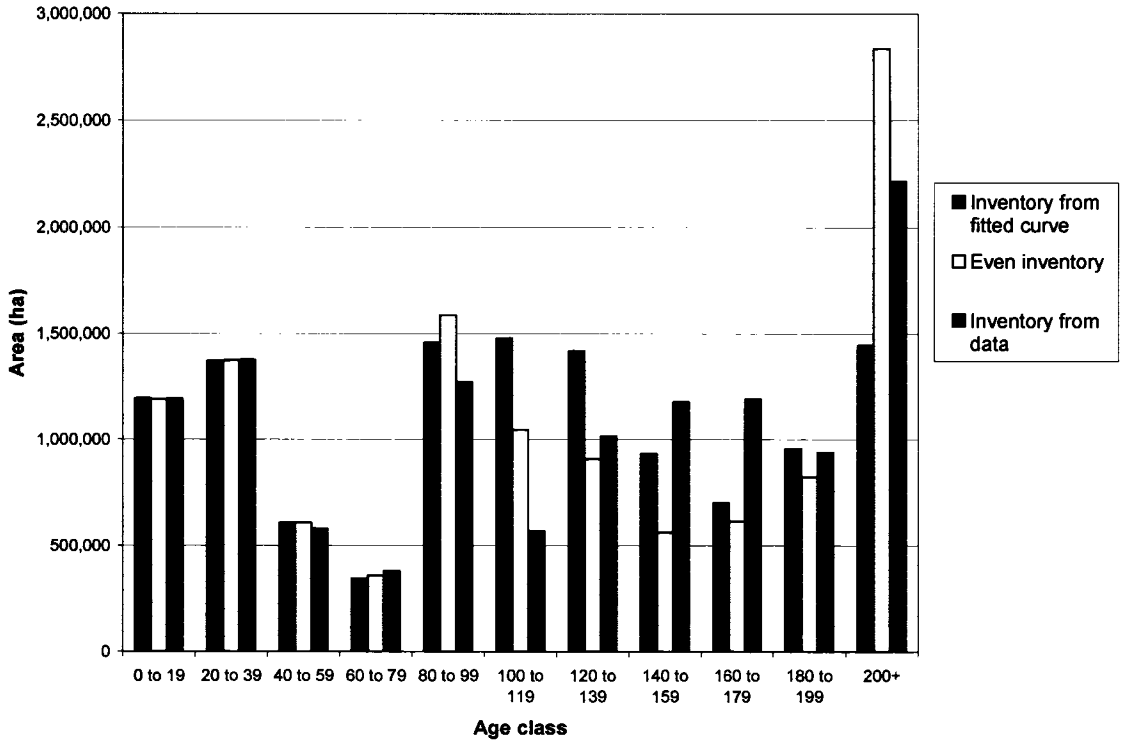


Figure 2.14. Age class distribution at the end of the simulation for three different initial inventory inputs. Concordance in the stand area of in younger age classes reflects the effects of fire disturbance simulated during the model run. The abundance of older stands is a result of the relatively low fire frequency in the "no climate change" scenario.

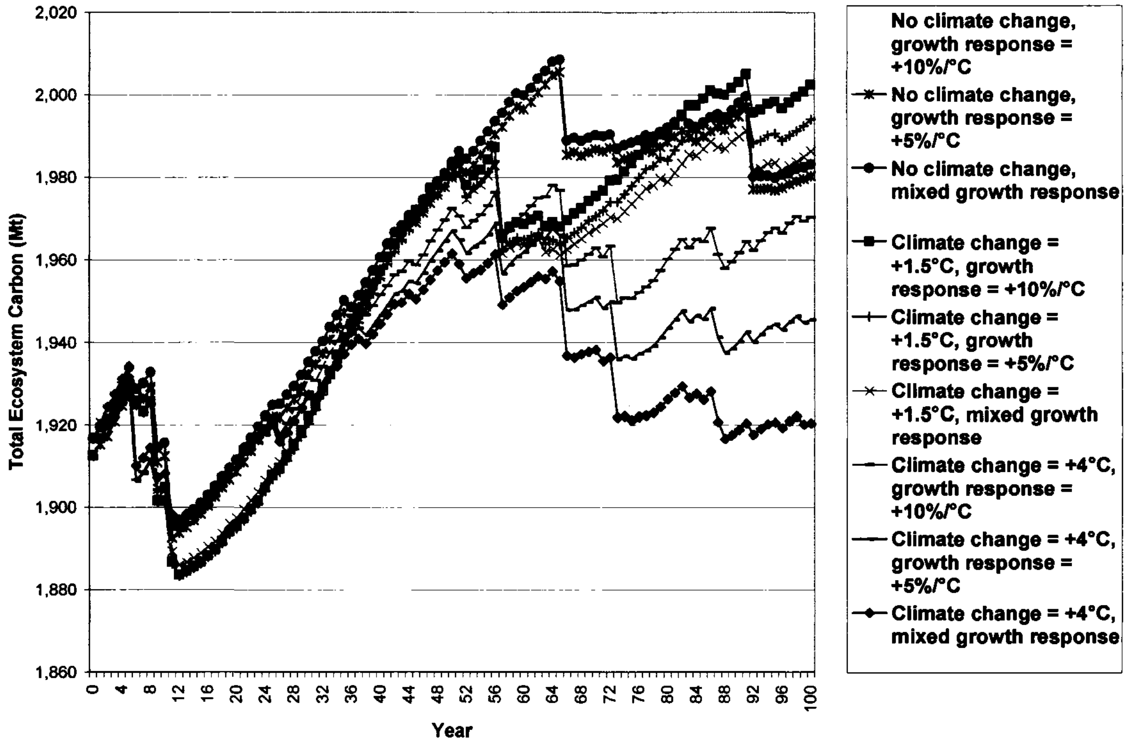


Figure 2.15. Sensitivity of model outputs to growth responses to climate change. For each climate scenario (no climate change, +1.5C, and +4C) model runs were performed using three alternative sets of growth curves (growth increases of 10% or 5% per °C increase across all species, or a 5% per °C increase in black spruce growth coupled with a 5% per °C decrease in white spruce growth). For both +4°C and +1.5°C climate change scenarios ecosystem carbon was clearly linked to both fire cycles and growth responses, with the greatest sensitivity to growth assumptions in the most extreme climate scenario.

Table 2.1. Cumulative sensitivity to growth responses to climate change after 100 years. Biomass carbon showed a positive response to temperature-induced increases in growth rates, but losses due to fire offset these increases in the +4C warming scenario. DOM carbon and total ecosystem carbon increased for all model runs.

	Biomass Carbon (Mt)	DOM Carbon (Mt)	Total Ecosystem Carbon (Mt)
Initial values (year 0), all runs	425	1,487	1,912
No climate change, mixed growth response	423	1,560	1,983
No climate change, growth response = +5%/°C	422	1,558	1,980
No climate change, growth response = +10%/°C	423	1,557	1,980
Climate change = +1.5°C, mixed growth response	432	1,554	1,986
Climate change = +1.5°C, growth response = +5%/°C	437	1,557	1,994
Climate change = +1.5°C, growth response = +10%/°C	442	1,561	2,003
Climate change = +4°C, mixed growth response	391	1,529	1,920
Climate change = +4°C, growth response = +5%/°C	405	1,540	1,945
Climate change = +4°C, growth response = +10%/°C	419	1,551	1,970

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CHAPTER 3

ASSESSING THE POTENTIAL FOR CONVERSION TO BIOMASS FUELS IN INTERIOR ALASKA:

CULTURAL CONSIDERATIONS, COSTS, AND CARBON CREDITS

ABSTRACT

In rural Alaskan communities, high economic, social, and ecological costs are associated with fossil fuel use for power generation. Local concerns regarding fuel prices, environmental contamination, and the effects of global climate change have resulted in increased interest in renewable energy sources. In this study I assessed the feasibility of switching from fossil fuels to wood energy in rural Alaskan villages in forested regions of Interior Alaska. Modeling results based on recent data on rural energy use, demographics, economics, and forest dynamics indicated that the installation costs of biomass systems would be recouped within ten years for at least 21 communities in the region. In addition, results showed that all but the largest remote communities in the Interior could meet all their electrical demand and some heating needs with a sustainable harvest of biomass within a radius of 10 km of the village. Marketable carbon credits may add an additional incentive for fuel conversion, particularly if U.S. prices eventually rise to match European levels. Biomass conversion also offers potential social benefits of providing local employment, retaining money locally, and reducing the risk of catastrophic wildfire near human habitation. This analysis demonstrated that conversion to biomass fuels is economically viable and socially beneficial for many villages across Interior Alaska.

INTRODUCTION

The excess carbon dioxide released into the atmosphere by the burning of fossil fuels is having measurable impacts on the earth's climate, with even more profound impacts likely in the future (Prentice et al. 2000; IPCC 2001; Karl and Trenberth 2003; Hansen et al. 2005a). Moreover, fossil fuels are a non-renewable resource with uncertain future prices and availability due to limited supplies and fragile international trade agreements. Thus, academic, industrial, and governmental researchers are increasingly exploring renewable sources of energy.

Potential sources of sustainable energy include solar, geothermal, hydroelectric, wind, and biomass. While each of these options has positive and negative attributes, biomass energy holds immediate promise because it is broadly available, fairly well developed technologically, and in some cases can be linked to benefit streams in addition to the production of energy. In the United States, interest in woody biomass as a fuel is increasing, as both an alternative fuel and a means of reducing fire risk near forested communities (GAO 2005).

The two primary obstacles that currently limit the use of woody biomass in the US are low cost-effectiveness and lack of reliable supply (GAO 2005). For example, the cost of producing electricity from woody biomass using current technologies in the US is currently 7.5 cents per kWh (kilowatt hour), whereas the market price for this electricity is only 5.3 cents per kWh (GAO 2005).

These obstacles might be overcome if selected communities can institute pilot projects that demonstrate the efficacy of biomass energy, provide a testing-ground for improvements, and at the same time enjoy immediate economic and social benefits on a local scale. I propose that the ideal locations for such pilot projects might be in communities with the following attributes:

1. Relatively small and self-contained with simple infrastructure
2. High current cost of power and/or heat
3. Proximity to sustainable supplies of woody biomass
4. Lack of social opposition to use of biomass fuel
5. Strong social impetus to mitigate global climate change
6. Interest in obtaining marketable carbon credits
7. Existence of other social and economic considerations that make biomass harvest and use a desirable option.

Many villages and towns in Interior Alaska fit all of these criteria. Rural Alaskans are disproportionately exposed to the effects of climate change, which is most pronounced at high latitudes (ACIA 2005), and struggle with rising fuel costs in a mixed economy characterized by high transportation costs. In rural Alaskan communities, mainstream fossil fuel technologies are prohibitively expensive. Large quantities of alternative fuels in the form of woody biomass (chiefly black spruce, *Picea mariana*) are available in this region, and the technology to use these fuels is relatively simple. Moreover, positive economic externalities may be realized through forest thinning or clearing, given the risks of forest fires to life and property, the direct costs of fire suppression, and the negative impacts of fire suppression on long-term ecosystem services. The advent of carbon trading markets in both the public and private sectors provide a source of additional revenue for alternative energy projects that could potentially tip the balance toward renewable energy sources (Duval 2004). Conversion of village diesel generation facilities to renewable energy sources is one way in which villages might partially mitigate climate change, earn tradable carbon sequestration credits, reduce fuel costs, reduce fire risk, and increase local autonomy, thereby reducing vulnerability to external social and economic change.

In many regions both in the US and abroad, immediate transition to alternate fuels is limited for economic, technological, or sociopolitical reasons. However, in much of Interior Alaska, economic drivers, governmental infrastructure, available natural resources, and social imperatives all point towards the viability of conversion to new energy sources. Rural Alaskan communities can be viewed as social-ecological systems. I suggest that fuel conversion programs could be implemented in such a manner as to have positive effects on these systems. I further suggest that Interior Alaska has the opportunity to provide leadership in this arena and serve as an example for other regions to emulate.

In this paper I analyze the feasibility and sustainability of potential biomass energy programs in rural Alaska by creating a social, biological, and political model framework within which I evaluate not only a wider range of financial costs and benefits, but also the interactions of ecological feasibility, social acceptability, community interest, and leadership commitment.

BACKGROUND: SYSTEM COMPONENTS

Energy Systems in Rural Alaska

Approximately 200 villages in Alaska have no connection to the electrical grid that links Alaska's largest communities. Prior to the 1960s, electricity was not available to most rural Alaskans (AVEC 2005). Now, these villages are generally supplied with electricity by diesel generators ranging from about 15 to 3100 kW in energy output (AEA 2000a). In total, 382,971,145 kWh of power were produced through diesel generation in Alaska in 2004, and 28,476,898 gallons of diesel fuel were consumed (AEA 2004). Many rural communities are part of regional cooperatives, including the Alaska Village Electrical Cooperative, Inc. (AVEC), which operates more than 150 diesel generators in 51 communities that run a cumulative 414,822 hours a year (AVEC 2005).

Because most rural Alaskan communities are not on the road system, fuel for these generators must be transported by barge or airplane. Thus, in most cases, fuel can only be transported during summer, and enough fuel to last a full year must be stored on-site (Colt et al. 2003). Maintaining this large storage capacity for fuel has posed significant environmental problems and incurred hundreds of million dollars of expenses (Colt et al. 2003; Duval 2004).

Due to the high costs of fuel transport and storage in rural Alaska, energy prices are extremely high. Consumers pay an effective rate of up to 35 cents per kWh in some regions. Less than half the total cost of electricity in rural Alaska can be directly attributed to fuel costs (Colt et al. 2003). Storage alone adds an estimated $1.50/gallon, due to capital expenses and spill response capability – which itself may add as much as 60 cents per gallon (UAF 2005).

Even in urban areas, electricity is more expensive in Alaska than in other parts of the country. In Fairbanks, the largest community in the Interior and Alaska's second-largest city, residential power costs over 11.6 cents per kWh, not counting additional charges (GVEA 2005), 35% more than the nationwide average cost of residential electricity (EIA 2005).

In rural areas, much higher costs occur despite substantial subsidies. The State of Alaska's Power Cost Equalization Program (PCE) provides assistance based on an algorithm that discounts costs by 95% between 12.0 and 52.5 cents per kWh (AEA 2004). Average residential rates without the subsidy would be more than 60 cents per kWh in some communities. Even so, the combined costs borne by consumers and the PCE program still do not account for a large proportion of the real costs of the system, due to funding from government grants, mostly for infrastructure. For small independent villages that are not AVEC members, these grants cover more than half (55%) of the real costs; for AVEC members, they cover approximately 26% (Colt et al. 2003). As the umbrella group for all village energy programs, the Alaska Energy Authority (AEA) administers and/or funds rural power system upgrades, the Power Cost Equalization (PCE) program, energy conservation and alternative energy development, the circuit rider maintenance and emergency response program, utility operator training, a bulk fuel revolving loan fund, a power project loan fund, and maintenance of AEA owned facilities. Although AEA has its own capital fund, recent capital project funding for bulk fuel storage upgrades and rural power system upgrades has come primarily from the Denali Commission, a federal-state partnership established by Congress in 1998 to provide critical utilities, infrastructure, and economic support throughout Alaska. It has been supplemented by other federal grants from agencies such as the Environmental Protection Agency (EPA) and the Department of Housing and Urban Development (HUD), as well as by State capital appropriations.

Rising fuel prices are likely to be the single greatest driver for a change from diesel-only systems. Diesel power generation is expensive in both direct and hidden costs. Among these are air pollution; problems with effective storage, resulting in soil and groundwater contamination from spills; spills during transport or transfer, resulting in larger-scale contamination and risks to humans and wildlife; risk of non-delivery of fuel under adverse conditions, resulting in loss of power; and dependency on the PCE program (Colt et al. 2003). There are approximately 1100 above-ground tank farms (each consisting of one or more tanks) in 161 remote villages in rural Alaska. A typical rural village has separate tank farms owned and operated by the City government, the tribal government, the village corporation, the local school, the electric utility, and other public or private entities. Up to 97% of these tank farms have serious deficiencies, including inadequate foundations, dikes, joints, and piping; improper siting near water sources; and rust and corrosion (Poe 2002; EPA 1999).

Biomass Investment and Technology

Developing village biomass projects is timely, given new interest and potential funding for wood energy in Interior Alaska. The Alaska Wood Energy Development Task Group, a recently formed coalition of federal and state agencies and other not-for-profit organizations, is now actively coordinating the State's efforts to increase the use of biomass for energy in Alaska. As of November 2004, the Task Group has been soliciting biomass energy project proposals from communities for funding with AEA earmarked funding. Currently AEA has budgeted $100,000 for wood energy activities ($84,000 USDOE and $16,000 AEA capital funds). During FY06, AEA expects to allocate an additional $349,000 in State and Federal funds to wood energy (AEA 2005).

Wood fuel has traditionally been converted into energy via open burning, fireplaces, and wood stoves. In traditional applications, the energy efficiency of biomass fuels for heating, cooking, and energy production is very low – in some cases as low as 10% (Kishore et al. 2004). However, biomass technology has improved over the past decade and has enjoyed success in other parts of the world, including Scandinavia and India. New biomass technologies allow for both more efficient energy conversion and – due to a hotter and more complete burn –greatly reduced emissions of particulates and carbon monoxide. Biomass fuels can include whole trees, cut firewood, chunk-wood, compressed sawdust pellets or briquettes, or gasified wood. These fuels can be used for electricity generation, heating, or a combination of both. Modern methods that offer greater combustion efficiency and lower emissions of air pollutants include combustion in a modern boiler/steam turbine system, direct wood gasification, or pyrolysis (Bain et al

1996). Although energy release is highly efficient in all of these systems, considerable energy is lost in converting that energy to electricity. Typically, the overall efficiency of a system that is only used to generate electricity is a mere 25-30% (Bain et al. 2003). However, much of the energy lost is converted to heat. If heat is also a desirable product, as is the case for most of the year in Interior Alaska, the boiler system can be configured for the simultaneous production of heat and electricity. More than fifty rural Alaska communities – or approximately 27 percent -- already have combined heat and power (CHP) systems (Crimp and Adamian 2001, MAFA 2004) and therefore have the infrastructure for heat and power distribution. Although system configurations vary widely, a preliminary assessment of the market indicates that 70 percent of rural Alaska communities could make cost effective use of combined heat and power systems (MAFA 2004).

Boiler systems are the simplest choice for biomass heat and power generation. In such a system, whole-tree wood chips or chunks are oxidized with excess air circulation, either in a stoker or a fluidized bed, and the hot flue gases released produce steam in the heat-exchange sections of a boiler. Some of this steam produces electricity via a turbine in a Rankine cycle, while the excess steam is used for heat (Bain et al. 2003).

Wood gasification and pyrolysis are potentially 30-40% more efficient than direct combustion, require less water, and result in cheaper costs per KWh, but generally involve more complex operation and maintenance requirements and newer and less proven technology (Scahill 2003). Wood gasification is the process of heating wood in an oxygen-limited chamber to a temperature range of 200-280°C until volatile gases including carbon monoxide, hydrogen, and oxygen are released from the wood and combusted (Bain et al. 2003). Several methods of gasification exist; however, updraft gasifiers are the simplest and most reliable (Scahill 2003) and thus the only type considered in this analysis.

Carbon Markets

Although the United States is not a signatory to the Kyoto Protocol on Climate Change, and policy analysts predict that CO_2 reductions will not become mandatory in the U.S. in the near future (McNamara 2004), the ramifications of this international agreement, as well as the dialogue that led to its creation, have nonetheless altered the way in which U.S. carbon stocks and fluxes are likely to be managed in the future.

In signatory nations, long-term carbon sequestration has become a commodity that can be traded against carbon emissions based on a cap-and-trade system (McNamara 2004). Likewise, reduction of emissions from non-renewable sources (generally fossil fuels) can be traded against increases in other sectors. In January 2005, the European Union – including all 25 of its member states -- initiated the European Union Emissions Trading Scheme (ETS), a legally binding international trading market in greenhouse gas emissions. Russia, Canada, and Switzerland are working towards instituting parallel systems (Kirk 2004). The transferability of carbon credits has opened up international economic possibilities never before seen, although some parallels can be drawn to the successful mitigation of SO_2 pollution in the US through use of tradable pollution credits (CCX 2006).

Meanwhile, non-governmental markets have already appeared, even in non-signatory nations. In the US, the Chicago Climate Exchange (CCX) is currently the most viable carbon credit market (McNamara 2004). It is acting as a self-regulating voluntary market, administering the world's first multi-sector and multi-national emission-trading platform. By participating in trading through CCX, corporations, municipalities, and other institutions have made legally-binding commitments to reduce net emissions of greenhouse gases. Carbon emitters as well as credit holders are banking on future increases in the price of credits, due to either international agreements or state and local laws. By entering the market early, buyers are showing good will and environmental responsibility, as well as setting up relationships that may prove lucrative in the future (McNamara 2004).

Alaska has yet to participate in nascent carbon markets, although the passage into law of a bill promoting carbon credit research (Berkowitz 2004) demonstrates the state's interest in both climate change and carbon-credit trading. Some states and geographic regions are already making local commitments to reduce greenhouse emissions. For example, in August 2001 the New England Governors and Eastern Canadian Premiers signed a regional climate change agreement aimed at reducing greenhouse gas emissions to 1990 levels by 2010, and reducing emissions by 10 percent below 1990 levels by 2020. In order to meet the requirements of this agreement, participatory states are creating local control mechanisms. In California, Governor Schwarzenegger signed Executive Order S-3-05 in June 2005, dictating that the state's greenhouse gas emissions would be reduced to 2000 levels by 2010, to 1990 levels by 2020, and to 80 percent of 1990 levels by 2050 (Schwarzenegger 2005).

Under the rules of the Kyoto Protocol – which are often used as guidelines, even in non-signatory markets – tradable credits can be obtained in a number of different ways, including afforestation, reforestation, and conversion from fossil fuel use to carbon-neutral fuels. For the purposes of carbon accounting, biomass can be considered carbon neutral; although carbon is emitted when biomass is burned, forest regrowth should, over time, take up an equal quantity of carbon. However, because the time scales of emissions and absorption differ, the sustainability of the forests from which biomass is harvested must be certified. All emission reductions and tradable carbon credits must be monitored, verified, and certified by a third-party that provides both confirmation that the carbon exists and insurance that it will be sequestered for the duration of the commitment period. Marketable carbon offsets also require proof of additionality – an assurance that sequestration or emission reductions would not have occurred had the project not been implemented. Finally, projects must not lead to "leakage": emission increases in another sector that can be attributed to reductions in the credited sector (Innes and Peterson 2001; UN 1997).

In Interior Alaska, fuel substitution may hold the greatest promise for attaining marketable carbon credits. Unlike credits based on afforestation, reforestation, or increased forest stocking, fuel offset credits are not one-time credits; as more fossil fuel use is offset over time, more credits can be earned. In addition, biomass energy generation can theoretically be developed on a wide range of scales. Finally, as described above, fuel offsets may be possible within a framework that generates other positive outcomes in addition to reduction of carbon emissions.

Forest Ecology and Ecosystem Services

The ecological sustainability of any proposed biomass fuels project will be pertinent not only from the point of view of achieving certifiable forestry practices in order to verify carbon sequestration credits, but also from the perspective of maintaining other ecosystem services. Historically, naturally occurring fires in Interior Alaska have created a variegated landscape with multiple age-classes of forest succession (Dyrness et al. 1986), each of which provides different resources (e.g. berries, moose browse, cover for furbearing mammals, and habitat for woodland caribou). However, fire suppression around inhabited areas tends to decrease average annual area burned (Dewilde and Chapin in press), which over time will tend to increase average forest stand age and reduce this variability, while also increasing the risk of future fires. While harvest and fire do not result in identical post-disturbance trajectories (Rees and Juday 2002), harvest does offer a means of introducing age-class variability and reducing fire risk around communities.

Goals and Objectives

The purpose of this study is to assess the feasibility of switching from fossil fuels to wood energy in rural Alaska villages located in forested regions of Interior Alaska (Figure 3.1) that are not supplied with electricity via the railbelt (the centralized power grid connecting Anchorage, Fairbanks, and other relatively large communities). More specifically, the study's objectives were to:

1) create a quantitative ecological model of the footprint of potential biomass harvest for wood energy around Interior Alaska villages;

2) create a quantitative economic model of the short-term and long-term costs and benefits of switching from diesel energy to wood energy in these remote communities;
3) explore the effects of model input selection and model parameter uncertainty on model outputs;
4) qualitatively assess the effects of social factors on the feasibility of fuel substitution;
5) and examine potential feedback between ecological, economic, and social factors, and assess ways in which they might in combination affect the feasibility of wood biomass fuel use in Alaska villages.

METHODS

Ecological Feasibility

For selected Interior Alaskan villages I created a simple model to estimate the area required to supply aboveground tree biomass over a rotation length that would mimic natural fire cycles while reducing fire risk in communities, optimizing aesthetic and subsistence values, and protecting ecosystem integrity. The biomass required was calculated from input variables and model parameters selected based on published data. Input variables included village size; village per capita energy needs; and optimal harvest rotation length. Parameters internal to the model included forest cover, forest volume, predicted biomass growth curves, and energy outputs by harvest volume.

Model output was expressed as maximum travel distance to obtain wood fuel – in other words, the distance between a village and the perimeter of the circle circumscribing the area of sustainable yield necessary to meet the needs described by the input variables.

The general formula used was $\text{Dmax} = \sqrt{\dfrac{P \times Epc \times Eo \times R \times 0.01}{Bd \times Ad \times Ew \times Ee \times Fc \times \pi}}$

Where:

Dmax =maximum travel distance (km)

P=village population

Epc= per capita energy use (kWh/yr)

Eo = Energy offset (fraction of total energy use replaced with biomass energy)

R=Rotation length for forest harvest (years)

Bd = biomass density (t/ha) for black spruce at harvest age (green weight)

Ad = correction factor for converting green to air-dried wood (t air-dry/t green)

Ew = energy available from air-dried wood (kW/t)

Ee = electrical efficiency (fraction of gross heating value converted to electrical energy)

Fc = Forest cover (black spruce forest as fraction of total land area)

I obtained nominal model results for villages within the study area by using mean, median or generally accepted values as initial model parameters. Nominal parameter values were selected conservatively, so as to over-estimate rather than under-estimate the footprint of harvest for biomass fuels around any particular village. Likewise, parameter ranges were selected to represent a relatively broad set of possible outcomes. Because all model inputs and parameters were part of a single first-order equation, and because all variables were multiplicative, the sensitivity of the model to variability in each parameter depended only on the magnitude of the range of possible values for that parameter. However, some of these ranges were quite large, resulting in a substantial cumulative effect of parameter uncertainty. I examined the sensitivity of the model to uncertainty in both model inputs and model parameters by performing three hundred stochastic model runs – one hundred for each for minimum, mean, and maximum community sizes -- using parameter values randomly selected from within each parameter range.

Model inputs reflected known or predicted values for village sizes and energy usage based on Alaska census data and information published by the Alaska Energy Authority (AEA 2000a; 2002; 2004; ADCED2005; Table 3.1). Mean population for the communities I focused on was 106, with a range from 21 to 1439. I considered energy use at current levels, based on kWh generated rather than kWh actually used in order to account for inevitable waste. The mean value was 3758 kWh per capita, close to the

4000 kWh estimated by Colt (2003). Communities with the highest usage were similar to the US average of 10,000 kWh per capita (Colt et al. 2003).

Rotation length was also treated as a model input, since it depends on community preference. I assumed that communities would seek to reduce wildfire risk as a byproduct of their harvest strategy and that they would therefore only harvest mature black spruce stands (the most fire-prone landscape type). An 80-year rotation would allow for harvest in early maturity, while a 200-year rotation would yield trees in late senescence; very few stands older than 200 years can be found for any species in Interior Alaska (Yarie and Billings 2002). Thus, I bounded the range of inputs with these values. The nominal value was set at 110, just prior to apparent age-related and/or fire-related decreases in stand frequency (Yarie and Billings 2002; Hollingsworth 2004).

Across the Interior, black spruce stands account for approximately 44% of the landscape (Sharratt 1997). This was used as a nominal value, although the actual mean is likely to be higher due to undercounting of early succession stands that would be classified as black spruce in a later successional stage. Since villages in areas with less than 10% forest cover were not considered, 10% was set as the low value, and 75% was selected as an upper limit (Fitzsimmons 2003). Although forest cover approaches 100% in some regions of the Interior, land around villages often contains considerable areas of rivers and other wetlands, so a conservative estimate was chosen.

The energy value of dry spruce chips was bracketed within a relatively small range by different authors (Maker 2004; Somashekhar et al. 2000; Zerbin 1984), making my model relatively insensitive to changes in this parameter. Based on these estimates, I selected a nominal value of 8500 btu/lb (5480 kW/t), with low and high boundaries of 7780 and 8920 btu/lb (5018 and 5353 kWh/t). However, differences in moisture content substantially affect energy output, since in the case of high-moisture fuel some of the energy released by combustion is used to evaporate water (Table 3.2). Although many wood burner systems can be used with a wide range of fuel types and fuel moistures, air-dry black spruce was selected as the nominal fuel, due to the general availability of the species and the relative technological ease of air-drying as compared to kiln-drying.

Green black spruce has a moisture content (MC) of approximately 60% (Yarie and Mead 1982), while air-dried wood has approximately 12-15% moisture (Yarie and Mead 1982; Prestemon 1998). Although this figure may in some cases be lower In Alaska's dry climate, I assumed an air-dried moisture content of 15%, and thus a typical weight loss of 28% during the drying process, and a final gross heating value (GHV) of 85% of the oven-dry value. Boundary values for these parameters were set at 0% weight loss and 40% GHV for green wood (Table 3.2), and 31% weight loss and 90% GHV for wood at 10% moisture.

Average aboveground tree biomass (including the fresh weights of bole, branches, and foliage) for 80-year-old. 110-year-old, and 200-year-old black spruce stands in Interior Alaska are approximately 25 t/ha, 28 t/ha, and 10 t/ha, respectively (Yarie and Billings 2002). It is likely that the low value for 200-year-old stands reflects the result of slow growth on shallow saturated soils; such stands would be less than optimal for biomass fuel management. I selected 28 t/ha as both the nominal and the maximum value, and 10 t/ha as the minimum value.

I assumed a nominal efficiency of 28% for electrical production, with a range of 20% - 40%, based on the estimates shown in Table 3.3. Overall efficiencies for combined heat and power systems are significantly higher. However, I chose to focus on the feasibility of wood-fired electrical generation and thus treated heat energy as a positive externality.

Economic Feasibility

Rural Alaskan villages have mixed economies that include significant market and non-market components, and the costs of current village energy programs are borne not only by community members but also by external entities. Thus, in order to analyze the economic sustainability of potential fuel offset programs, I considered not only the costs and benefits of construction, operation, maintenance, fuels, employment and carbon sequestration credits for diesel versus biomass systems, but also circulation of cash income and non-cash commodities within communities, and the effects of subsidies. I examined economic feasibility based on published estimates and projections for:

1. village energy consumption
2. fossil fuel costs
3. non-fuel expenses, including fuel transport and storage and system maintenance
4. existing subsidies for fossil fuels, infrastructure, and maintenance
5. installation and maintenance costs for biomass systems
6. labor and mechanical costs for wood procurement
7. existing village economies, cash flows, and employment
8. current and potential future prices for carbon credits

I created a quantitative model incorporating the above components in order to assess whether fuel conversion would be likely to have a positive economic outcome for each individual village, and over what time period initial investments in biomass infrastructure might be recouped.

The model input was the biomass generation capacity installed. Parameters internal to the model included diesel prices, non-fuel expenses, installed diesel capacity, and actual kWh generated -- for which I used published village-by-village values -- as well as installation costs for biomass systems and annual operation costs for biomass systems. For these parameters I determined nominal values based on mean, median or generally accepted values from the literature. Nominal parameter values were selected conservatively, so as to over-estimate rather than under-estimate the costs of fuel conversion. Likewise, parameter ranges were selected to represent a relatively broad set of possible outcomes.

I examined the sensitivity of the model by randomly selecting parameter values for key variables (diesel price, biomass system installation costs, annual biomass operation and maintenance costs,- and carbon credit prices) from the full range of uncertainty expressed in the literature. Using these random values, I analyzed the results of 10 stochastic model runs under complete replacement of diesel systems and 10 runs under a partial replacement scenario for each of the 31 villages for which adequate data were available (a total of 620 model runs).

The general formula used in the economic sub-model was:

$$Y = \frac{CapitalCosts}{AnnualSavings}$$

$$= \frac{CapitalCosts}{(AnnualCostsOffset) - (AnnualBiomassCosts) + (AnnualCarbonCreditValue)}$$

$$= \frac{Ic \times El}{[(Ao \times De \times Dp) + (NFo \times NFc)] - (Bg \times Bc) + (De \times Ao \times Cc)}$$

Where:

Y = years to pay back investment

Ic = installed cost of a biomass power system, per kW generation capacity

El = electrical load (total biomass capacity installed, in kW)

Ao = actual offset, in Kwh (based on relationship between installed biomass capacity and mean electrical load)

De = diesel efficiency (gallons of diesel fuel per kWh generated)

Dp = diesel price ($/gallon for diesel fuel)

NFo = non-fuel offset (fraction of non-fuel diesel generation costs offset by biomass, based on relationship between installed biomass capacity and mean electrical load)

NFc = non-fuel costs (total $)

Bg = Biomass energy generated (kWh/yr)

Bc = Biomass energy costs ($/kWh)

Cc = carbon credits available due to fuel offset ($/gallon fuel)

Total capacity installed in each village, total annual energy use in each village, and much of the data on non-fuel costs and existing costs and funding sources for power systems was available through state Department of Community and Economic Development budget requests (Poe 2001, 2002) budget reports (Alaska 2001, 2002), the UAA Institute of Social and Economic Research (UAA 2003), and the Alaska Energy Authority (AEA 2000a, 2002, 2004, 2005). These parameters were incorporated in the model as given. Average fuel prices were based on 2004 figures, despite the steep rise in prices over the following year. However, I assessed the sensitivity of this parameter within the range of -50% to +150% in order to account for this volatility.

As a nominal model input, I assigned biomass capacity installed in each village a value equal to the mean electrical load for that community. Under this assumption, existing diesel systems would be at least partially retained and maintained in order to meet peak loads, while allowing biomass systems to run at full capacity for much of the time. In the communities I assessed, mean load was only 8%-29% of installed capacity (Appendix), demonstrating over-capitalization that it would probably not be necessary to replicate with biomass systems. Load profiles are not available for most rural Alaskan communities. However, available information from six villages of varying sizes shows combined daily and seasonal variation yielding peak loads that are approximately twice mean loads and three-fold greater than minimum loads (Devine et al. 2005). Installation of biomass generation capacity greater than minimum loads would result in some percentage unused capacity; at a capacity equal to mean loads this would represent approximately 30%, while at a capacity equal to twice peak loads this would represent approximately 60% (Figure 3.2). Diesel fuel costs would be directly offset according to the number of kilowatts actually generated by the biomass system, while non-fuel expenses for diesel systems would be offset according to the total capacity replaced. Thus, I estimated that continuous operation of biomass systems at mean load levels would offset 60% of the village's diesel fuel use, but only reduce by 25% non-fuel expenses associated with existing systems (eg fuel storage, O&M, and replacement of diesel generators). In order to assess the sensitivity of the model to my assumptions, I compared the results with a model run in which biomass generation capacity replaced 100% of existing capacity (eliminating diesel systems altogether), and another run in which biomass generation replaced only 50% of mean loads, thus replacing 40% of diesel fuel use and 10% of non-fuel expenses (Devine et al. 2005).

I compiled estimates of capital costs for purchase and installation of biomass systems from a range of available sources (Table 3.4). In order to present conservative approximations in estimating feasibility of fuel conversion, and in order to allow for the potentially higher costs of installation and operation in remote Alaskan sites, I use mean of the authors' high-end estimates, \$1849/kW, as the nominal value in my model. For the purposes of sensitivity analysis I considered the range of values between the minimum published value (\$980/kW) and 125% of the maximum published value (\$2500 x 1.25) = \$3125.

Operation and maintenance costs for wood-powered systems are difficult to accurately estimate, since they depend on location, wages, ease of fuel procurement, mechanization of harvest, and ease of maintenance. In rural Alaska, travel costs and lack of local technical expertise would be expected to drive up the costs of system maintenance. However, this is already the case for diesel systems. Small-scale relatively non-mechanized methods for gathering and chipping wood might increase labor costs per ton of fuel, but the ready availability of both wood fuel and labor might partially balance these effects. I estimated the cost of fuel procurement based on actual costs of clearing and thinning projects in rural communities (Table 3.5) (Hanson 2005 pers. comm.; BLM 2005; Lee 2005). In all cases, local crews were used, and the work was labor-intensive and low-tech. Although these projects did not entail using the harvested wood for electrical generation, they did include manual disposal through piling and burning or chipping, as well as overhead and equipment costs. Translating these costs into equivalent energy costs resulted in a mean or nominal value of \$0.16/kWh (rounded up to \$0.17/kWh). In order to provide a more conservative estimate of feasibility in my sensitivity analysis and avoid reliance on a potentially anomalous value, I raised the lower end of this range to four times the costs recorded for Stevens Village (to \$0.12/kWh), and rounded the upper limit \$0.28. Projected costs are similar to estimates of between \$0.06 and \$0.20 per kWh (mean = \$0.16/kWh) noted by various sources (Haq 2002; Anonymous 2001; Bain et al 2003; USDA and USFA 2004; Scahill 2003) (Table 3.5). This is substantially less than the real cost of diesel power in most villages.

I gathered information on village-by-village fuel use, energy use, fuel costs, and subsidies primarily from annual statistical reports on the Power Cost Equalization

Program (AEA 2000a; 2002; 2004) and Alaska Electric Power Statistics for 1960-2001 prepared by the Institute of Social and Economic Research at UAA for the Alaska Energy Authority, the Regulatory Commission of Alaska, and the Denali Commission (UAA 2003). Some of these data have already been shown in Table 3.1; the full dataset appears in the Appendix.

I estimated model parameters for the value of carbon sequestration credits by gathering data on existing markets in the US and in Europe and calculating the tons of carbon offset for each 1000 gallons of diesel replaced by biomass fuel. The estimated value of these credits covers a wide range, due to market fluctuations and future uncertainty. Prior to the Kyoto Protocol taking effect in signatory nations, the trading price of carbon was typically slightly over one dollar per metric ton. In 2005, prices fluctuated around the two dollar mark, and I used a value of $1.90 in my analysis, despite the fact that 2006 values have spiked as high as about $4. While the international agreement had no direct effect on U.S. markets, it appears to have had an indirect effect (McNamara 2004). However, the prices of these voluntary credits remain far below the prices for verified emissions reductions in signatory nations. On the ECX, the European trading market, prices rose from approximately €8 ($9) at the beginning of 2005 to almost €30 in July 2005, and in August 2005 settled back down to about €20 ($24) (McCrone).

Carbon credits represent a benefit stream from outside the village economy, with a value additive to all other benefit streams. I analyzed the potential value of the credits that could be obtained on a village-by-village basis, based on the number of tons of diesel offset, as determined by village energy use and biomass capacity installed (model input) (Table 3.6). Although derived via different algorithms, my results, which estimate a total of 32,609 t of CO_2 emissions from diesel power generation in rural Interior forested communities, are congruent with those obtained by Duval (2004), who estimated a total of 274,263 metric tons of CO2 emissions for all PCE communities, with 52,047 of these tons from "forested Alaska." My somewhat lower figures for forested Interior Alaska reflect the fact that some rural forested communities are in the southeastern or south-central parts of the state, which are not considered in my analysis.

Social Feasibility

Analysis of social feasibility was primarily qualitative rather than quantitative, and included assessment of:

1. Existing social infrastructure related to village electrical utility management and funding, fire prevention, and biomass harvest
2. Threshold requirements (make-or-break factors needed within a particular community or at a broader scale, e.g., a minimum level of local technological expertise)
3. Existing institutional barriers to change
4. Potential positive social feedback (e.g., autonomy, employment)
5. Potential negative social feedback (e.g., reactions to system quirks or failures)
6. Lessons learned from existing biomass projects in rural Alaska

Although funding for village power systems is provided to a large degree by state and federal subsidies via AEA programs, ownership and operating responsibility for many of these projects is placed entirely with local grantees (Poe 2002). Thus, I assumed that most ultimate decision-making would take place at the village level, although financing, training, infrastructure, and technological expertise might all come from farther afield.

In addition, I drew information from past and ongoing projects with goals and objectives similar to those proposed in this study. These include wood fuel projects such as the existing boiler at Dot Lake and the proposed biomass system in McGrath (Adamian et al.1998; AEA 2000b; Crimp and Adamian 2001; Crimp 2005); other alternative fuel projects such as wind-diesel hybrid systems (MAFA 2004; AEA 2005; Devine et al. 2005); and fire prevention efforts that include forest clearing (Hanson 2005; Putnam 2005).

Several fuels treatment projects aimed at reducing the risk of catastrophic wildfire have already taken place in village settings, under a combination of local leadership and assistance from entities such as the Alaska Department of Natural Resources Division of Forestry and Tanana Chiefs Conference. The immediate costs of these projects were noted in Table 3.6. However, in order to further ascertain the impacts of these efforts at the village level, I spoke with Doug Hanson of DNR (2005 pers. comm.) and Will Putnam of TCC (2005 pers comm.). In particular, I questioned the importance of local hire; the role of key leaders, elders, or crew bosses; and the relationship between fire crews, harvest crews, and local opinions regarding fire protection.

Although for the purposes of the economic sub-model I calculated costs and benefits irrespective of the impacts on different funding sources and beneficiaries, analysis of benefit streams was necessary for a more in-depth understanding of the social sub-models. Thus, I qualitatively assessed the current discrepancy between the real costs of power and the costs borne by consumers; the potential impacts of shifting funding and changing subsidies; and the potential economic value of local jobs generated by the harvest of biomass fuels. My analysis was based on data on existing sources of funding for Alaska rural energy projects (Table 3.7); data from the Power Cost Equalization Program (Appendix) (AEA 2004); and financial information from past forest clearing projects (Table 3.5)

Results

Ecological Feasibility

Using nominal parameter values and a forest rotation length of 110 years, the maximum travel distance required to collect enough mature black spruce to meet average electrical loads (thus supplying approximately 60% of total village power) ranged from 1.1 km to 12.8 km. (Table 3.8).

With the exception of the two largest communities, Tok and Galena, which have regional and local road systems, respectively, the maximum travel distance was calculated to be 6.2 km or less, a distance easily reachable by snowmachine or four-wheeler, allowing for relatively low-tech harvest using chainsaws and a portable chipper. Larger communities might still find biomass fuel conversion an attractive option if they are located in regions with sufficient forest cover or road access, and if per capita electrical use remains modest. Even if 100% of village energy needs were supplied by biomass, the maximum travel distance for communities of up to 600 inhabitants would be no more than 8 km. (Figure 3.3). Selecting a rotation length of 80 rather than 110 years only modestly reduces the maximum travel distance (Figure 3.4), since shorter rotations are correlated with lower biomass densities. However, increasing the rotation length to 200 years greatly increases the harvest area and travel distance, due to both the longer return interval before stands can be harvested again, and reduced spruce biomass per hectare in older stands.

Model sensitivity analysis using randomly selected parameter values from within each parameter range yielded a distribution of results for each of three village sizes (Figure 3.5). For a village of 21 residents, no model runs yielded a maximum travel distance of over 3.8 km; the mean was 1.7 km. For a village of 106 residents, the range was 1.5 to 10.7, with a mean of 3.9 km. The distribution of results was broadest for the largest communities with a single outlier at 39.3 km. The remainder of the range fell between 5.5 km and 27.5 km, with a mean of 14.2 km.

Economic Feasibility

Due to missing data, not all economic calculations could be performed for all selected communities. For some villages, data were missing for fuel costs, non-fuel expenses, or energy generated (Appendix), making it impossible to include these communities in model results. Thus, my results reflect a subset of forested-off-grid villages in the Interior. However, in addition to obtaining village-specific results, I was able to explore general relationships between village size, village accessibility, and economic feasibility.

For many of the communities in this analysis, total annual operating costs for electrical generation would be lower if all or part of the village's diesel power were converted to a biomass-fueled system (Table 3.9). Only Tok, Northway, and Koyukuk show consistently negative results; however, since Tok and Northway are both accessible via the Alaska Highway, one of the state's major thoroughfares, they may be considered anomalous as compared to more remote villages accessible only by minor roads or by rivers (major or minor) (Appendix). Other communities -- including Evansville and Bettles, Eagle, Minto, Tanana, and Tetlin – show benefits from conversion to wood fuel under some conditions but not under others, depending on the scale of the biomass generation capacity installed.

When the added benefit stream of potential carbon sequestration credits is added to the potential annual savings gained by biomass fuel conversion, wood-fired electrical generation becomes more favorable. Even without taking carbon credits into account, 23 communities show a payback period of less than 25 years for the initial capital investment of installing a biomass electrical generation system adequate to meet mean electrical loads (Figure 3.6). The projected time before a net positive economic balance is reached without carbon credits ranges from a mere 0.7 years for Lime Village and 1.3 years each for Stony River and Red Devil, to 11.7 years for Tanana and 15.8 years for Minto. If communities were able to sell carbon offset credits at 2005 US prices, the payback periods for Tanana and Minto would drop to 11.3 and 14.9 years, respectively. At European carbon prices, these figures would dip to 7.8 and 9.2 years. Villages for which it would take longer than 25 years to recoup the investment and communities for which the benefit stream is negative are not shown in this figure. However, both McGrath and Fort Yukon, two of the larger communities analyzed, show a payback period of less than 25 years when carbon credits are taken into consideration, but not when carbon credits are not included.

It should be noted that, although villages for which data are absent have been necessarily omitted from this analysis, these communities should not be assumed to have a poor cost-benefit balance from potential biomass projects. In general, communities not accessible via a major road showed positive results based on biomass generation at mean load levels (Figure 3.7). This relationship was particularly robust for communities with fewer than 100 residents.

For many communities, my model placed biomass conversion close to the economic break-even point when nominal parameters were used. Stochastic model runs using randomly selected parameter values from within broad possible ranges yielded results that encompassed both positive and negative economic outcomes for almost all the villages analyzed, for both partial biomass conversion (Figure 3.8) and total replacement of diesel power (Figure 3.9). Only 8 villages – Aniak, Central, Lime Village, Manley Hot Springs, Red Devil, Sleetmute, Stony River, and Takotna – showed net annual savings on operation and maintenance (O&M) costs for all ten model runs under both scenarios. However, only two communities – Fort Yukon and Tok – yielded unfavorable results in 50% or more of model runs for replacement of mean load capacity. Only Tok showed negative results half or more of the time when total system replacement was considered. As the largest community with the greatest power usage, Tok also yielded the broadest range of potential annual costs or savings.

The ranges used in this analysis included installed costs between $980 and $3125 per KW; annual operation and maintenance costs for biomass systems between $0.12 and $0.28 per kWh; carbon credits between $0 and $222 per 1000 gallons of fuel offset; and fuel prices between 50% and 250% of 2004 prices. It should be noted, however, that 2006 prices are already close to 200% of 2004 prices in many areas (Demarban 2006a). If 2006 prices were used as a baseline, model runs would become consistently favorable in almost all communities.

When capital costs for biomass system installation was also considered as a random stochastic variable and results were calculated for expected project payback time, results showed a similar pattern (Figure 3.10). Six of the eight villages for which all model runs yielded annual savings also showed a payback time of less than 25 years for all model runs. Only Tok showed a consistently poor ability to recoup the investment costs associated with biomass conversion, although other communities, including

Evansville and Bettles, Fort Yukon, Koyukuk, Minto, and Northway yielded mixed results.

Social Feasibility

My qualitative analysis of the potential social role of biomass fuel conversion in rural Interior Alaska yielded a conceptual map of where wood fuel might fit into village economies (Figure 3.11). Harvest of biomass fuels would provide local jobs, which in turn would bolster the local cash economy by recirculating money within each village. In contrast, payments for fossil fuels represent a monetary flow out of communities. Currently, economic multipliers in village economies are small. Income from carbon credits would create a cash flow into the community from an outside source – something that is often in short supply in rural Alaska. It should be noted that fire is linked to many aspects of community wealth, in both monetary and subsistence categories. Thus, natural forest succession, protection of life and property, local wages, and subsistence foods are all linked through the presence – or absence – of fire on the landscape.

Analysis of the impacts of subsidies and grants on village energy choices revealed a substantial gap between the real costs of electrical power and the prices being charged to consumers (Figure 3.12). Moreover, the real costs of village power comprise a substantial proportion of village income, ranging from 7.1% to 70.0% (Figure 3.13). Because I have included the electricity used in shared facilities such as washeterias, schools, and offices in my totals, my figures are much higher than those for household use only (Colt 2003). In reality, however, the discrepancy between realized costs and real costs may be even larger, due to hidden (off-book) costs covered by transfer payments other than those made via the Power Cost Equalization Program. These include government-funded construction and upgrades, many of which were listed in Table 3.7. Such off-book-costs account for roughly 25% of the real costs of power (Colt 2003), but are not accounted for in my economic analysis.

The gap between real and realized costs has negative social ramifications, creating disincentives for locally-based efficiency improvements, sustainable community planning, and innovative use of capital (Colt 2003). Even if biomass fuels can be shown to be an option that is feasible in a given community, village residents may lack the necessary economic incentive to catalyze change. Moreover, the small population base of most villages has in the past proven to be an obstacle to reliably securing the necessary human resources for governance, operation, and maintenance of utilities (Colt 2003). On the other hand, although government entities may have a financial incentive to promote change and may have the necessary technical expertise and human resources, they may suffer from bureaucratic inertia and lack of social impetus. Based on the financial power wielded at higher levels of governance and the social power contained within communities, there are potential advantages and disadvantages associated with both top-down or bottom-up approaches to managing potential village biomass projects (Table 3.10)

At the state and federal levels, grants and other sources of funding are available to cover startup costs, and technical expertise is available for design and implementation work, including funds specifically allocated to renewable energy and alternative power (AEA 2005). Most of these funds would likely be channeled through the Alaska Energy Authority, as detailed in Table 3.7.

The advantages of the infrastructural assistance and funding available through AEA give rural Alaska a potential edge over rural communities in less developed nations, where capital and technological inputs are more uniformly scarce. Even in India, a nation with a stronger economy than many developing nations, lack of financial support for technology improvements has been cited as the primary reason for failure of an early attempt at instituting a small-scale biomass energy project (Kishore et al. 2004). A national-level analysis in India showed that biomass gasifiers 20 to 200 kW in capacity could entirely meet rural electricity needs (Somashekhar et al. 2000). Some demonstration projects have proven relatively successful (Somashekhar et al. 2000) while others have not (Kishore et al. 2004). There are several reasons for project failure, including subsidized power available from the existing grid; extremely low purchasing power among village residents; and poor technology for burning biomass other than wood chips (such as rice husks and other plant residues.) (Kishore et al. 2004). Since Alaska's villages are largely removed from the power grid, have greater cash flows than rural Indian communities, and have wood as the primary source of potential biomass, these problems are unlikely to be applicable.

In addition to providing funding and know-how, governments may be the most effective managers of some aspects of on-the-ground efforts. Some degree of centralization and top-down effort are predicated by the tiny size of some of the communities in question. For example, specialized skills such as boiler design and installation and engineering of combined heat and power grid systems would not be found in every community of 50 to 100 individuals.

However, direct management from the state or federal level is rife with potential problems. The same remoteness that makes the cost of diesel fuel in villages so high also demands that village power and heating systems be internally rather than externally managed whenever possible. Cultural considerations bolster this assertion. Village residents, most of them Alaska Natives, strongly prefer local control of village affairs (Putnam 2005, Hanson 2005).

Not only is local autonomy culturally preferable, it is also likely to be crucial for the long-term viability of biomass projects. State and federal officials are unlikely to be knowledgeable concerning important details such as interpersonal dynamics in the community, traditional use in the area around the village slated for harvest, and local concerns regarding fire risk. For example, during community studies preliminary to the installation of a biomass energy system in the village of McGrath, residents expressed concerns about the technical and economic feasibility of the project; the impacts of increased wood harvest on subsistence activities, aesthetics, and future wood supply; and overall system complexity (Crimp and Adamian 2001). Alaska Natives are often suspicious of solutions derived by governmental groups that are perceived to be part of the problem, and without community support, trust, and buy-in, programs instituted by outside entities are doomed to failure (Reiger et al. 2002).

In addition, local residents are likely to be able to provide realistic assessments of what type of employment would be considered desirable, and on what time scale it might be undertaken. For example, wood harvest, chipping, and transport might be shared informally among several individuals, and might be timed not only to coincide with adequate snowpack for easy transport, but also to fit in with seasonal subsistence activities and other seasonal employment. In most communities, gathering wood fuel is already part of subsistence activities; community members would be best equipped to decide how and when to expand fuel collection, and how to pay individuals for the wood they gather. Since fuel gathering would be coupled with fire prevention, and because fuel collection would be most likely to occur in the winter via snowmachine rather than in the summer fire season, existing fire crews would be an obvious choice of labor force. Hanson (2005) notes that fire crews were involved in fuel clearing projects in Healy Lake, Tanacross, and Stevens Village, and that these groups generally work well together and are actively interested in fire protection. However, he also commented that work crews vary, and that having a good crew boss or leader is crucial to success.

Village councils, local light and power cooperatives, and Native corporations have greater power to implement projects than do individuals. For example, these entities are eligible for state or federal grants such as those being made available through the Alaska Wood Energy Development Task Group. These grants, however, are being channeled via AEA. At an intermediate level of governance, organizations such as AEA, AVEC, and other regional light and power cooperatives have the potential to help link the resources of governmental agencies with the resources of communities. These organizations have already taken a lead in proposing, funding, and implementing alternative energy projects (AVEC 2005; AEA 2005). AVEC has thus far focused on wind and hydro-electric power, since many of its customer communities are coastal. AEA has taken a lead in biomass demonstration projects, including installation of a wood-fired boiler in Dot Lake, and a proposal for a larger system in McGrath. AEA has garnered funding for such projects from the state and federal level, but is implementing them using criteria that take into account local needs and local capacities.

In the long run, a combined approach seems likely to provide the greatest resilience to the system. Power sharing and co-management are ideas that are starting to take hold in a range of rural applications and are likely to be appropriate in an Alaskan context (Reiger et al. 2002). For example, although overarching assessments of fuel supply and demand around a village might be performed by forestry professionals, annual harvest areas might be chosen by local village councils, based on community preferences.

Based on the above information, I identified the following barriers and thresholds to change:

Barriers

- The majority of AEA funding is traditionally allocated to existing system components, not to renewable energy or new technology startup
- In some cases, state or AEA capital funds are designated for programs such as PCE and bulk fuel revolving loans, which create negative economic externalities favoring the status quo.
- Many power cooperatives are managed regionally, not at the village level
- No forest certification system is in place whereby carbon credits could immediately be secured (although the potential for development of such a program exists within either the Alaska Department of Natural Resources or native corporation programs such as the Tanana Chiefs Conference Forestry Program).
- Failure by the United States to sign onto binding climate-change agreements may keep carbon credit prices an order of magnitude lower here than overseas.

Thresholds

- Existence/absence of human capital in the form of participatory individuals within village, particularly village leaders who are willing to advocate for a biomass program, fire crews or other individuals actively interested in employment and fire prevention, and one or more crew leaders who can take responsibility for follow-through.

- Existence/absence of social capital in the form of necessary skills within village, and/or the willingness of system operators to receive training in new technologies.
- Formation of effective cross-level collaboration, particularly between AEA (the likely funding agency and potential overarching project manager); village electrical companies or cooperatives (the likely applicants for funding and local managers); and individuals employed on the ground at the village level.

Discussion

The transition to renewable energy sources is constrained by a number of economic, social, technological, and political factors. These include start-up costs for research and new infrastructure; social inertia and risk aversion; inadequately developed technologies; lack of availability of all energy sources in all regions; and artificially low costs of existing fossil-fuel systems due to subsidies, lack of accounting for economic externalities, and current infrastructure. Nevertheless, my results indicate that even with conservative assumptions for ecological, economic and social parameters, conversion to wood biomass energy is likely to be a feasible and attractive option for many communities in Interior Alaska. A successful fuel-conversion program must fulfill the social, economic, and ecological needs of the system as a whole (Figure 3.14).

Based on my model, the communities likely to show the greatest ecological feasibility for biomass conversion are those in the small to medium size categories. Only the largest communities – those with populations over about 300 -- potentially lack adequate wood resources for complete fuel conversion within an easily accessible radius. This pattern runs counter to the trend whereby other services such as schools, clinics, and airports are more cost-effective in larger communities, leading to governmental pressure towards consolidation of small villages. Ironically, many villages have shrunk due in part to the high costs of fuel (Debarman 2006a).

The greatest economic feasibility is demonstrated by villages with the highest benefit/cost ratio, which tend to be those not easily reached by either road or river networks. For these villages, even high estimates of costs for fuel systems show an advantage over existing high costs for fuel transportation and storage.

Social feasibility, because it is so dependent on individuals, has yet to be determined on a village-by-village basis. However, it is likely to be greatest in communities with strong leadership, close ties to the land and its resources, and a core group of individuals – perhaps an existing fire crew -- willing and able to work consistently on fuels harvest and associated tasks. These requirements tend to point towards medium or larger communities in remote areas.

Villages that fit both the ecological and the economic criteria include Alatna & Allakaket, Anvik, Central, Chuathbaluk, Circle, Crooked Creek, Grayling, Healy Lake, Holy Cross, Huslia, Kaltag, Lime Village, Manley Hot Springs, Minto, Nikolai, Nulato, Red Devil, Shageluk, Sleetmute, Stony River, and Takotna. In the smallest communities in this group the presence or absence of strong leadership and willing workforce would be particularly critical in determining the success of conversion. For example, Takotna lists zero unemployed individuals from its 29 residents over the age of 16 (Appendix). On the other hand, Aniak and Tanana also show a positive benefit/cost ratio, but have populations above 300. Projects in these communities would have to be more cautious regarding wood supply, harvest area, and overall energy use, or might optimally be based on only partial conversion to wood fuel. Meanwhile Evansville & Bettles, Koyukuk, and Tetlin easily met ecological criteria but were on the borderline in the economic analysis. Fort Yukon, McGrath, Northway, and Tok all showed mixed results. These four communities are all either much larger than the mean, or located on a readily accessible transportation corridor, or both. Although biomass conversion projects may be feasible in these locations, additional factors would need to be taken into account, including the possibility of procuring wood from slightly further afield (via road or river), and the effects of biomass conversion on the larger and more complex economies of these communities. Finally, inadequate data were available to fully assess potential feasibility for Beaver, Dot Lake, Galena, Hughes, and Ruby.

My analysis was intentionally conservative, and may therefore have underestimated potential advantages of conversion to biomass fuels. For example, the 200-year forest rotations used in my sensitivity analysis are far longer than would likely be considered by communities seeking fire protection and habitat revitalization, and my estimates for biomass per acre, forest cover, and carbon credit prices were relatively low, while my estimates for biomass system installation costs were relatively high. Perhaps the greatest undercounting of potential system benefits stems from the fact that, although I assumed that installed systems would provide both power and heat, I accounted for only the savings afforded by replacing the existing power supply. Although heating could in most cases only be provided for centrally located buildings, the savings afforded would likely be substantial in communities that already have infrastructure to support combined heat and power distribution, and worth assessing even in those that do not. Including heat as a resource increases estimates of biomass generator efficiency from approximately 28% to 68% (Table 3.3). Even if less than half of this additional benefit stream could be effectively captured, it would increase the overall energy realized by more than 50%. An increase in system benefits of this magnitude would make almost all fuel conversion options economically attractive. Another potential source of error may stem from the fact that off-book expenses associated with current diesel systems were not considered, although they are likely to account for approximately 26% of total costs (Colt 2003). Finally, all estimates were made using 2004 fuel costs, which are substantially lower than current costs (Demarban 2006a;b). Fuel costs may continue to rise, and federal and state subsidies may shrink or disappear. The incentives for fuel conversion at the village level are highest when fuel prices are highest, but lower fuel costs might trigger the removal of state subsidies, since state revenues are almost entirely dependent on oil prices. These changes would make fuel conversion increasingly appealing – including, in many cases, conversion of 100% of generation capacity rather than partial conversion.

In addition to the sources of uncertainty explored in this analysis, other factors could affect the feasibility and desirability of biomass conversion programs. New transportation corridors might lower the costs of fuel transport in some areas. Additional local employment opportunities might drive up local wages, thus raising harvest costs or reducing the potential workforce. Payback on capital investments could be affected by inflation, deflation, or rapid changes in interest rates.

On the other hand, grant money such as that available through the Wood Energy Task Force, the Denali Commission, or AEA's Wood Energy Development Program could help jump-start projects, and might make infrastructure costs less of a concern. New technology might reduce the installation and operation costs for wood gasifiers below the range predicted, or international turmoil might cause fuel prices to skyrocket above predicted values. Carbon credit prices would eventually rise to match current ECX prices, even in the US, if new binding international agreements are reached. Moreover, if fire on the landscape is perceived as an ever-increasing threat, and if state and federal firefighting resources become strained, then forest clearing might become more socially desirable and/or financially lucrative in its own right.

Many of these potential changes or surprises would tend to increase the economic viability of fuel conversion. However, model uncertainty not only means that economic outcomes are ambiguous for many villages, but also that social feasibility is uncertain. Thus, pilot projects offer the next step in testing feasibility. Such projects would help to validate my model, test technology under new conditions (e.g. remote, cold climate), provide positive lessons that could be incorporated into future projects, and provide experience regarding errors to avoid.

When ecological, economic, and social parameters are considered in conjunction with one another, a pattern of hurdles and benefits emerges (Table 3.11). While many of these have been addressed in my analysis, others can only be truly tested through use of real-life project implementation.

Two existing pilot projects in Interior Alaska demonstrate the feasibility of wood biomass systems and the efficacy of employing combined heat and power capabilities. The first, a wood-fired boiler used to heat and power eight residences and the washeteria in the 37-person community of Dot Lake, is already operational. The second, in

McGrath, has not yet been completed, but is slated to include a combined heat and power system based on continued use of diesel with a wood boiler providing additional energy to the system.

Dot Lake is not a typical Interior village, since it is on the road system. As a result, diesel fuel in the community is far less expensive than in some villages, and my calculations show a strongly negative incentive for biomass fuels conversion. Nevertheless, Village Council President Bill Miller estimates that the village saves $6,500 to $13,000 per year using the wood-powered system (AEA 2000b). However, wood prices in Dot Lake are not likely to be equivalent to prices in more remote villages, since in Dot Lake wood has not been harvested solely as fuel. The boiler operates on wastes from nearby timber operations, which can be easily transported via road.

In McGrath, the option selected appeared economically preferable to three other possibilities: the status quo (all diesel); a wood boiler powering only the school; or a more comprehensive wood system, with diesel remaining as the back-up fuel (Crimp and Adamian 2001). Crimp and Adamian (2001) also noted that the cost-effective use of biomass was highly dependent on the availability of inexpensive wood wastes; costs would be expected to rise sharply when roundwood harvest was required to operate the facility. However, at the time the analysis was done, it was assumed that the cost of `bulk diesel would remain static at $1.54/gal. In reality, prices have risen sharply, increasing by 25 percent between 2003 and 2004, an additional 35 percent between 2004 and 2005 (Bradner 2005), and even further in 2006.

As previously described, the potential income from sale of carbon credits from Interior villages would be roughly $62,000 annually at 2005 market prices. In very small villages, the totals would be less than $300 per year. Even in larger communities, these sums represent only a very small percentage of the funds that would be necessary to operate and maintain combined heat and power systems of any kind. However, in some cases, these sums are enough to tip the balance towards biomass fuel conversion. If the value of carbon credits in the US ever rises to meet world standards, perhaps due to future international agreements, the additive value of these credits could become a significant part of the cash economy at the village scale.

CONCLUSION

Given the combined drivers of rising fuel prices, ongoing climate change, increasing fire risk, and social pressures favoring fossil fuel independence, many communities may soon consider shifting to alternative fuels. The incentive of earning tradable carbon credits has added to potential benefit streams, and the monetary gains of participating in carbon markets may increase tenfold or more in the long term if the United States eventually implements programs congruent with those being used by Kyoto Protocol signatory nations.

In rural Alaska villages, economic conditions make fossil fuel use unusually expensive, while social conditions favor autonomy and local employment. Ecological conditions are likely to allow for harvesting a sustainable fuel source in a matter that enhances rather than detracts from ecological resilience, due to the complex relationship between fire, forest succession, forest resources, fire suppression, and human settlements. Biomass fuels are likely to increase the long-term social and ecological resilience of village communities to externally-driven changes, including fluctuations in fossil fuel prices due to state, national, or international policies; variability in Alaska's economic outlook, which might in turn impact subsidies; and changes in fire risk and fire management, driven by climate change and by state and federal fire budgets.

For all of these reasons, Interior Alaska village communities are in a position to be at the forefront in developing biomass fuels programs. Villages selected based on my combined social, ecological and social model would almost certain reap benefits from the transition. In addition, due to the existence of substantial economic and political infrastructure at the state and federal level, Alaska's rural communities are in a position to serve as pilot projects and leaders in a global movement towards rural biomass power.

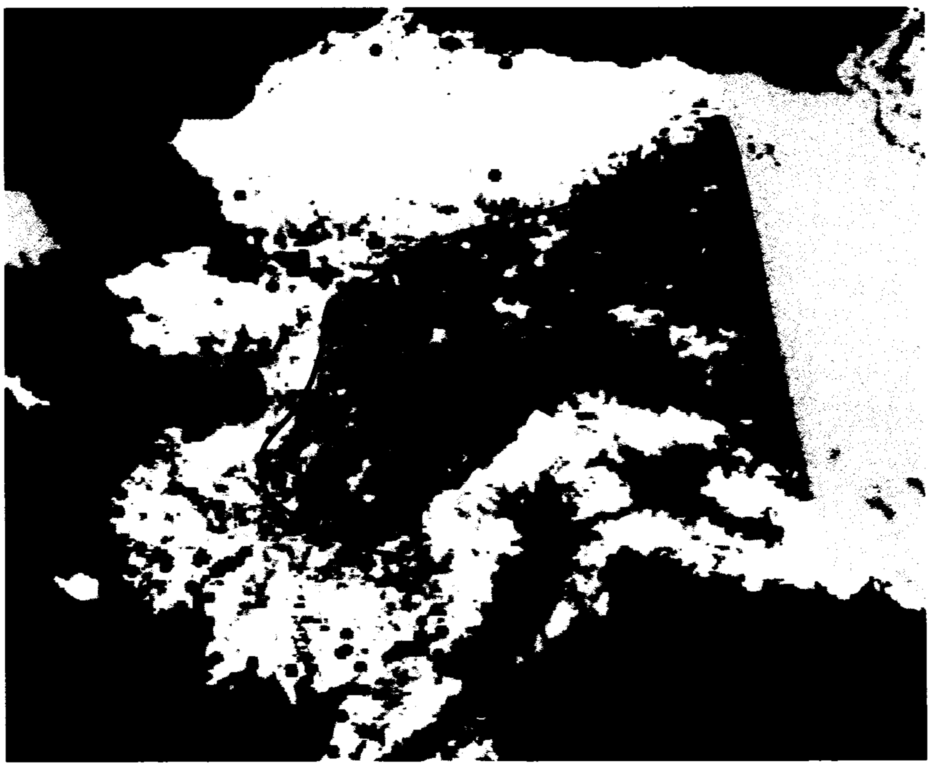

Figure 3.1. Remote Alaska communities. About 90 communities (represented by dots) lie in forested regions (green shaded area). Approximately half of these are in the Interior region considered in this study (roughly demarcated by black line). (adapted from Crimp and Adamian 2000).

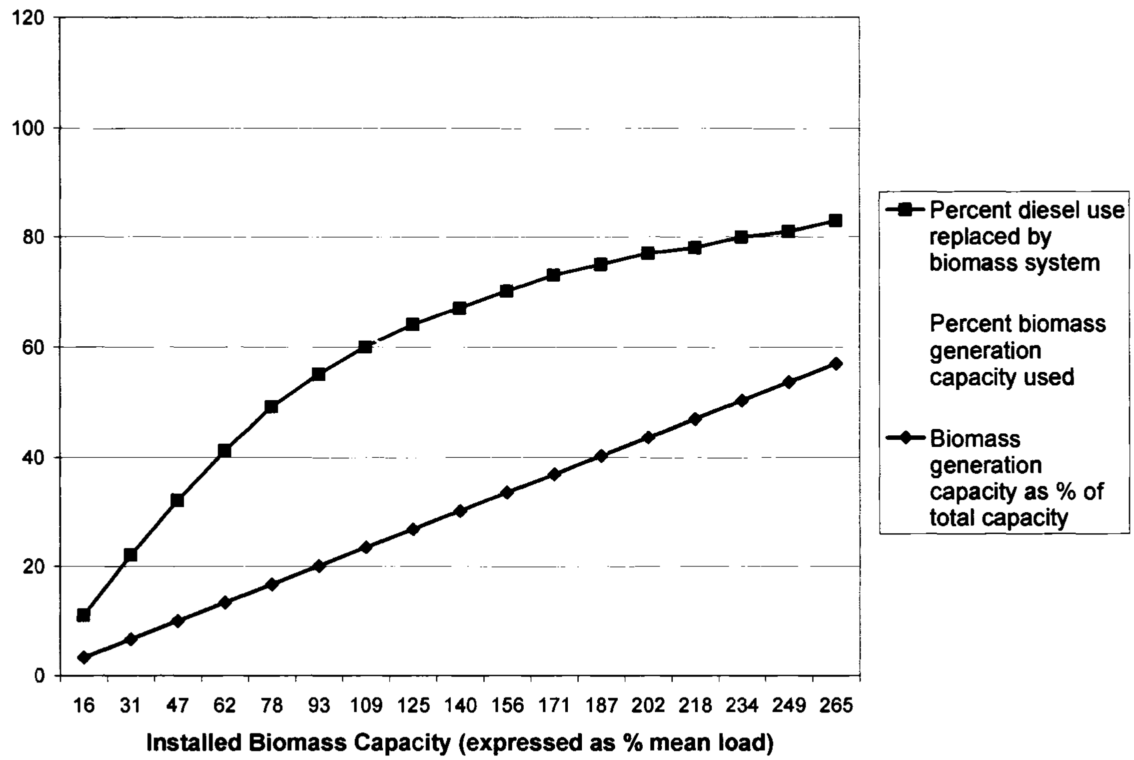

Figure 3.2. The relationship between biomass generation capacity, diesel fuel savings, and non-fuel expenses. Due to daily and seasonal variability in energy demands, total system capacity is designed to greatly exceed average loads (adapted from data on substitution of diesel systems with wind power, Devine et al. 2005).

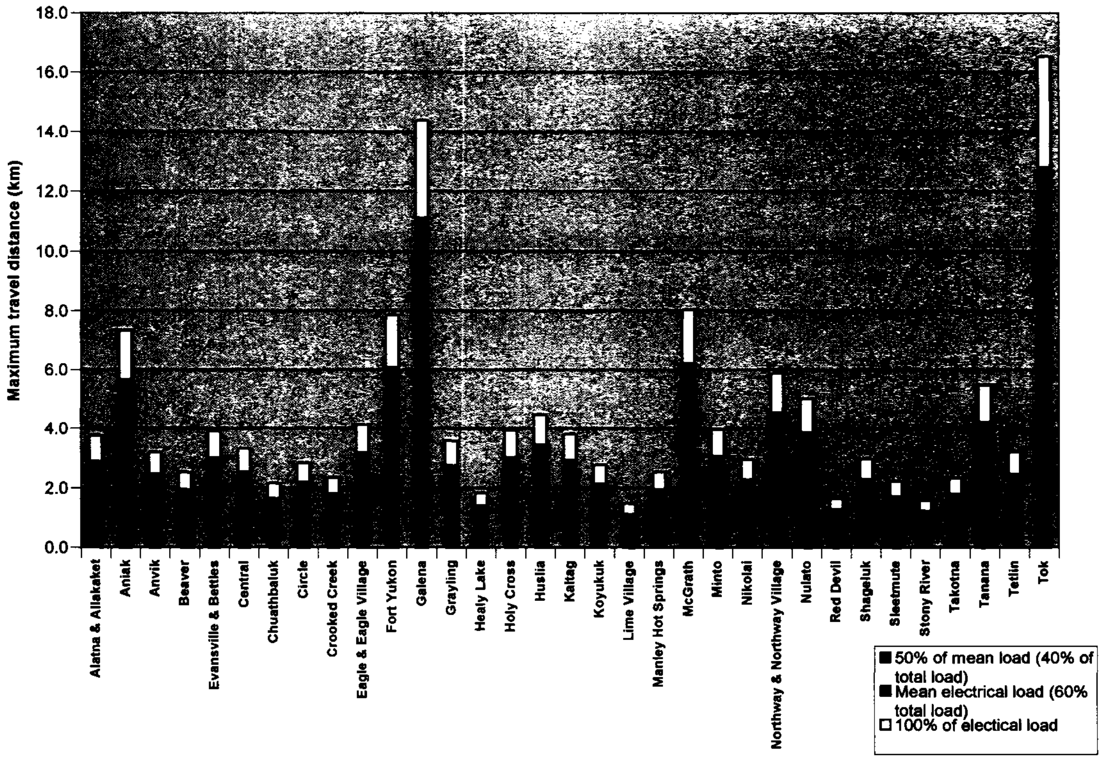

Figure 3.3. Maximum travel distance according to the percentage of village energy needs met by biomass fuels. Model outputs estimate sustainable harvest of black spruce for energy generation. If installed biomass generation capacity is equal to 50% of mean loads, approximately 40% of the community's electrical demand will be offset. At a capacity equal to mean loads this rises to 60%. All data assume 110-year forest rotations.

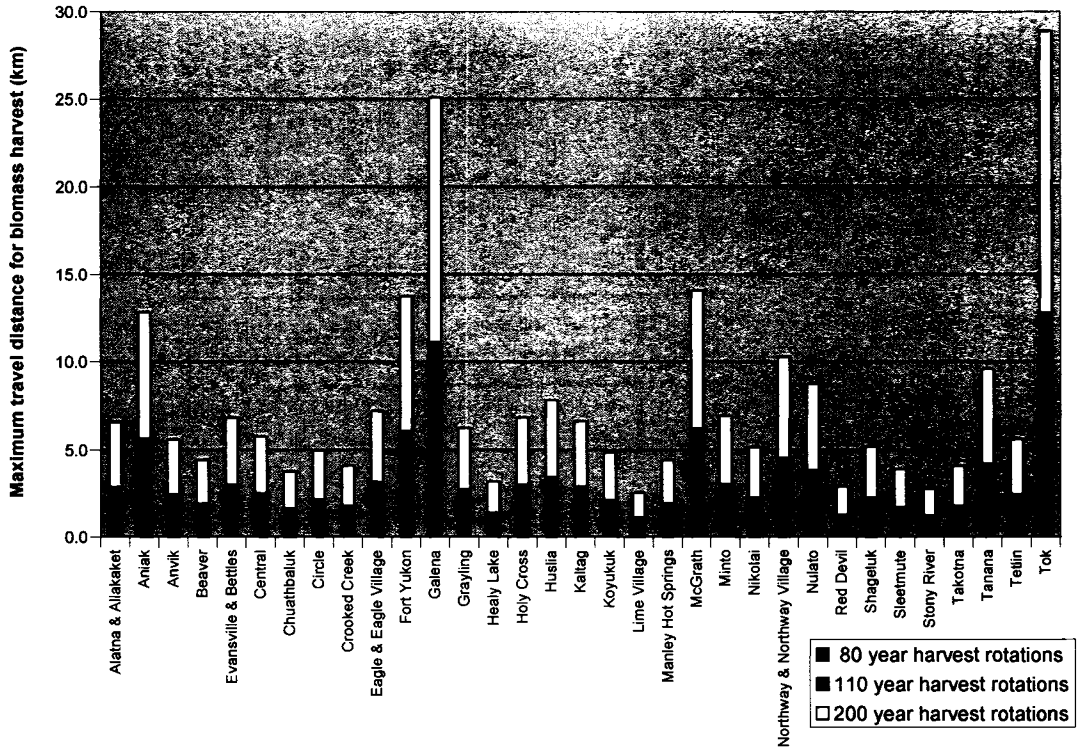

Figure 3.4. Maximum travel distance for sustainable harvest of black spruce for energy generation according to selected harvest rotations. Rotation lengths of 80, 110, or 200 years are shown. Black spruce biomass density per hectare increases between 80 and 110 years, and decreases between 110 and 200 years (Yarie and Billings 2002), resulting in a steep increase in travel distance with long rotations.

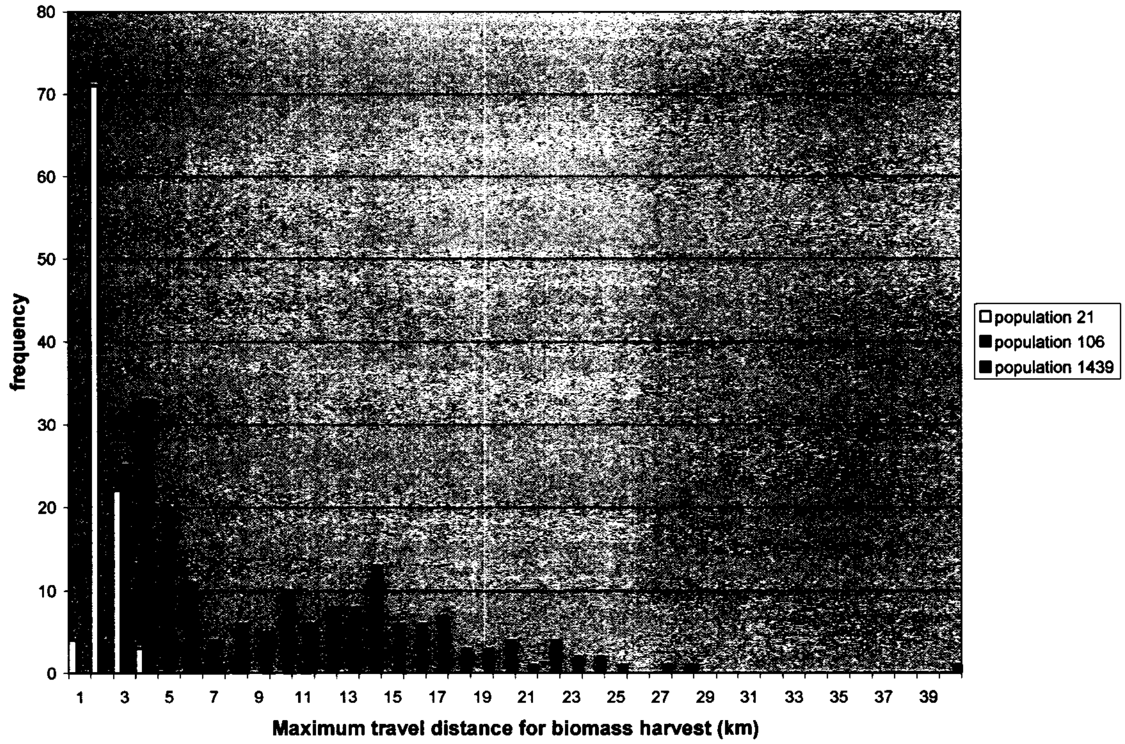

Figure 3.5. Distribution of results for 100 stochastic model runs for each of three village population sizes. In each model run, values for rotation length, biomass density, energy by moisture content, energy per ton, forest cover, and electrical efficiency were randomly selected from within broad accepted ranges.

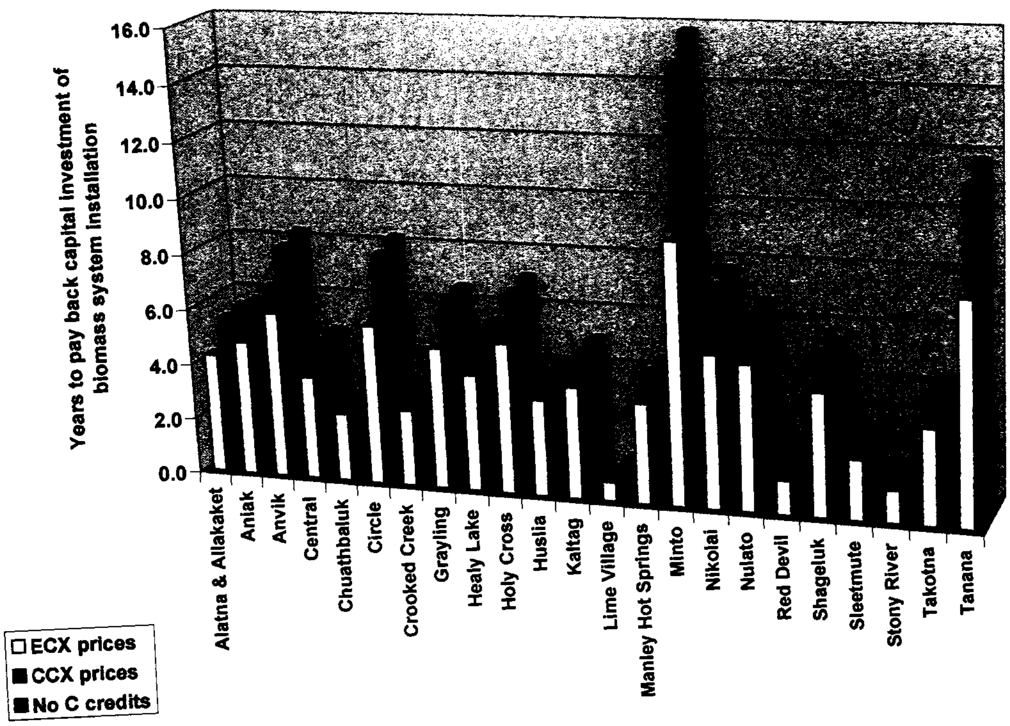

Figure 3.6. Years necessary to recoup an investment in wood-powered electrical generation capacity equal to mean electrical loads. For each selected village, three carbon-credit trading scenarios are shown: one in which no carbon credits are sold, one in which available fuel-offset credits are traded at current Chicago Climate Exchange (CCX) prices, and one in which credits are traded at current European Climate Exchange (ECX) prices.

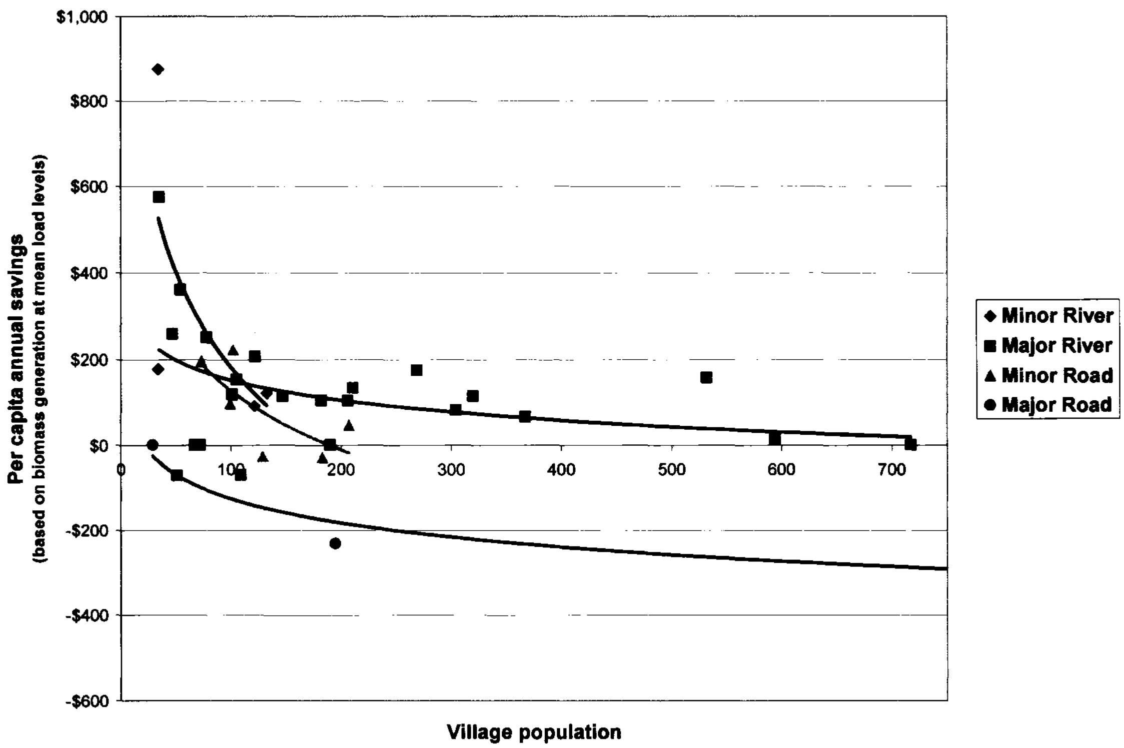

Figure 3.7. Per capita savings by village size and accessibility. Logarithmic regression curves are fitted to four categories of accessibility. Only those villages that can be reached on major roads show consistently negative results for replacement of fossil fuel with biomass fuels at mean load capacity. For all villages, smaller population size is correlated with greater per capita benefits from fuel conversion.

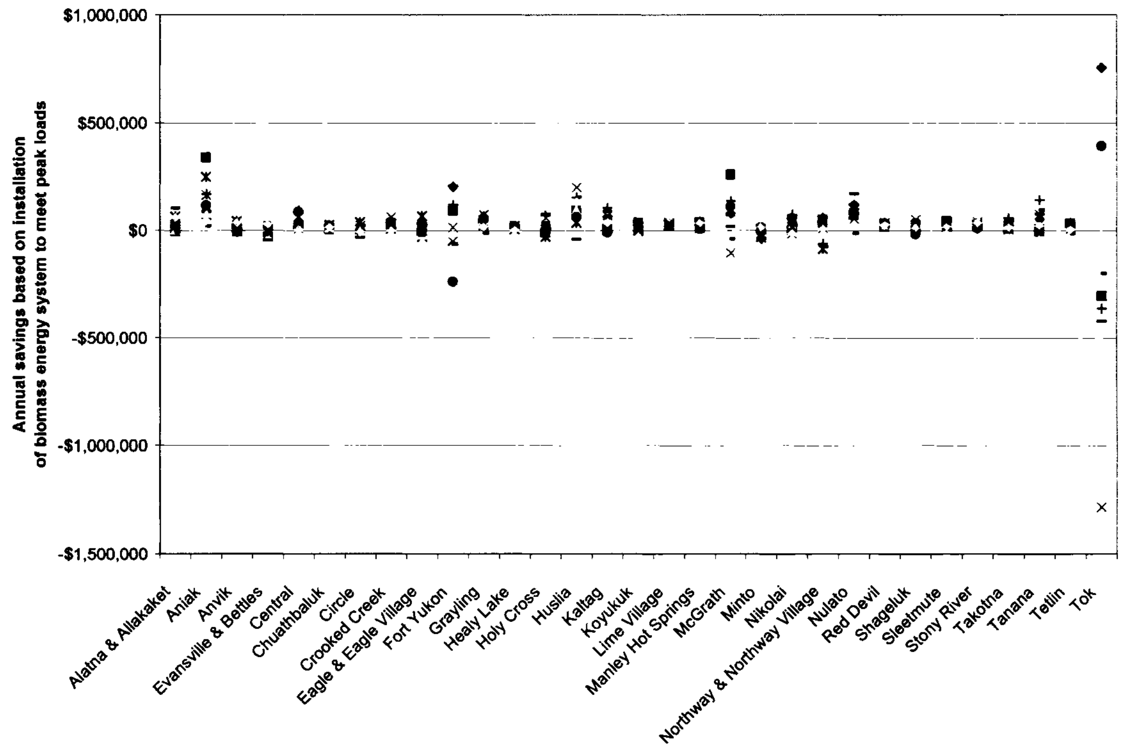

Figure 3.8. Sensitivity analysis for partial replacement of diesel systems with biomass electrical generation. Data points show the results of ten model runs using parameters randomly selected from within broad possible ranges.

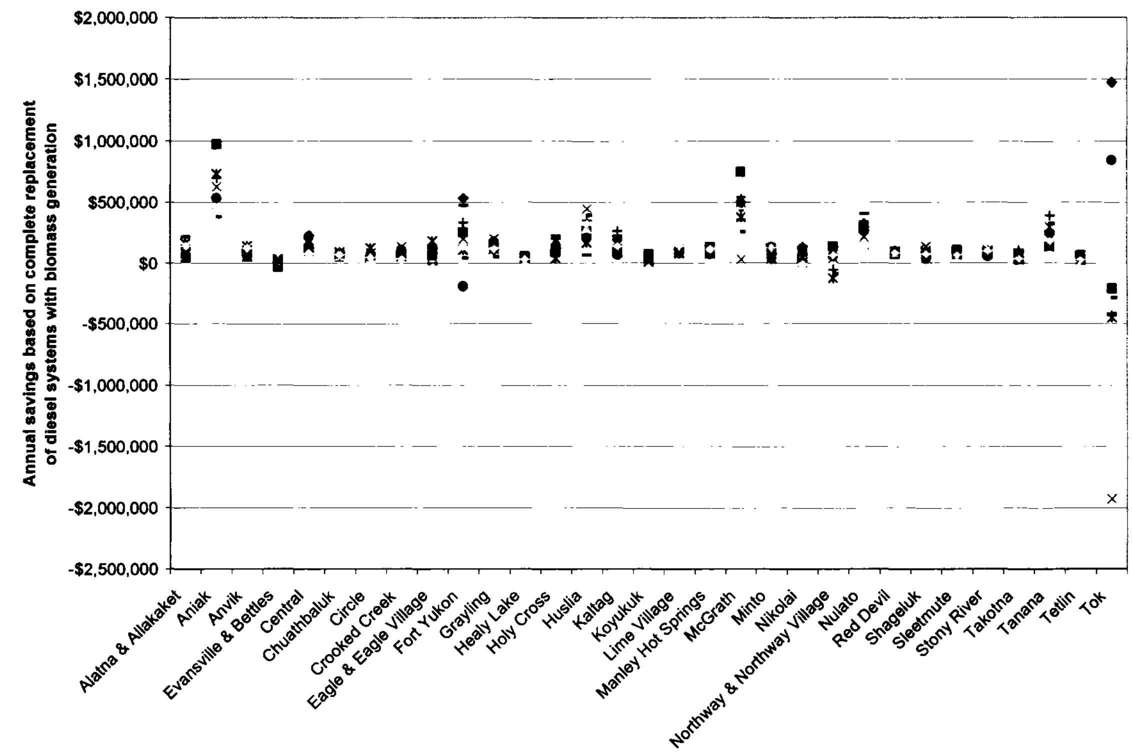

Figure 3.9. Sensitivity analysis for total replacement of diesel systems with biomass electrical generation. Data points show the results of ten model runs using parameters randomly selected from within broad possible ranges.

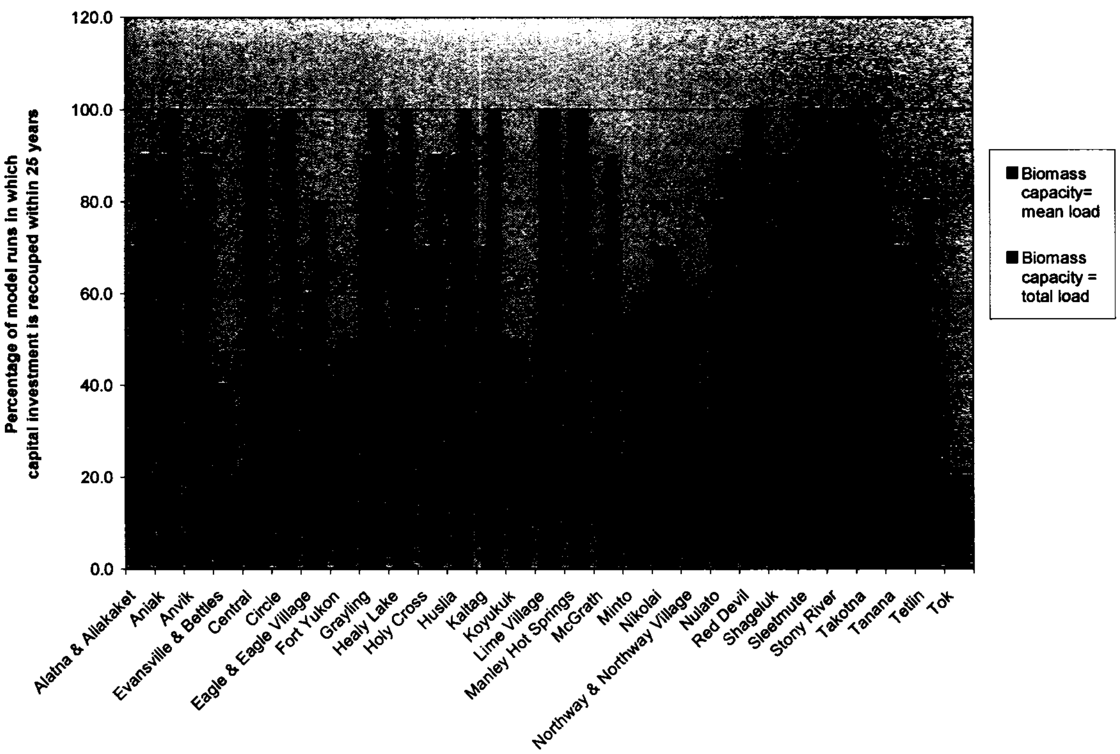

Figure 3.10. Sensitivity analysis of time necessary to recoup capital investment. For each village, the graph illustrates the percentage of stochastic model runs for which randomly selected parameter values yielded a payback time of less than 25 years.

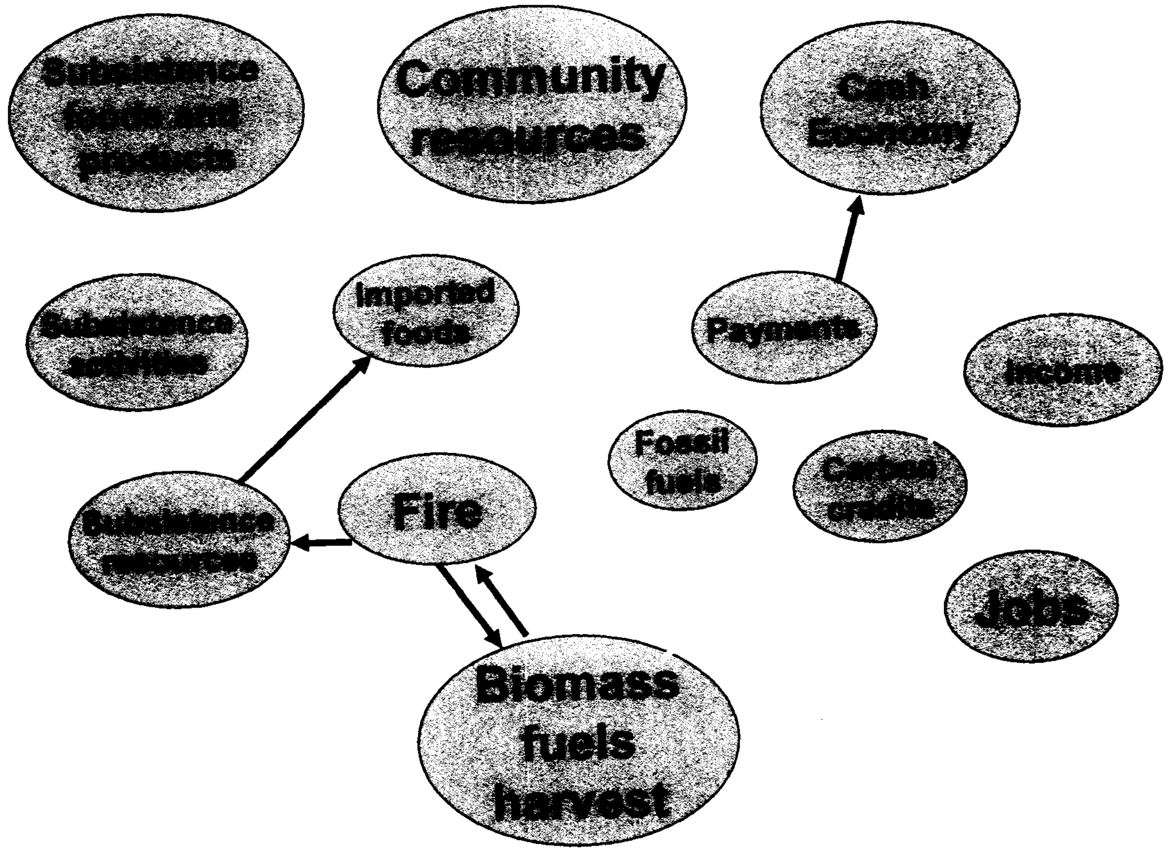

Figure 3.11. A conceptual model of economic feedback interactions. Village market and non-market economies are potentially linked to biomass fuels programs. Black arrows indicate positive effects and red arrows indicate negative effects. Note that fire can have both positive and negative impacts on subsistence resources, depending on time scale.

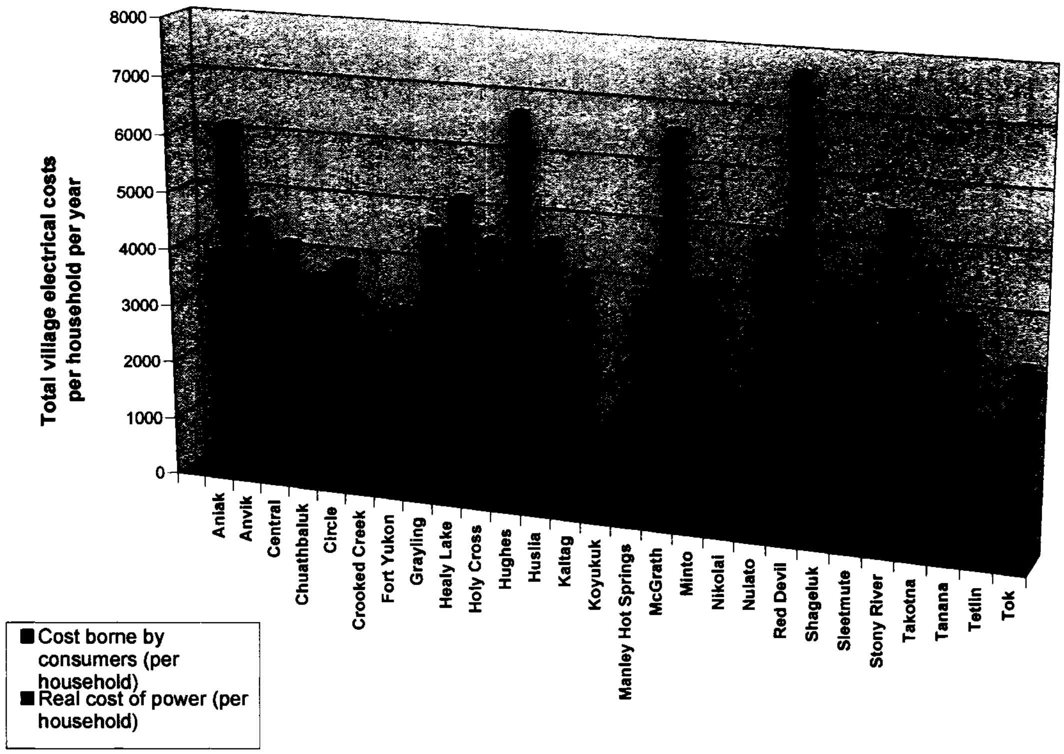

Figure 3.12. Village electrical costs, expressed on a per-household basis. These figures include costs incurred for electrical use in private homes as well as in shared facilities such as schools, tribal offices, and washeterias. The discrepancy between the costs borne by consumers and the real cost of power is covered by government funding, primarily via the Power Cost Equalization Program.

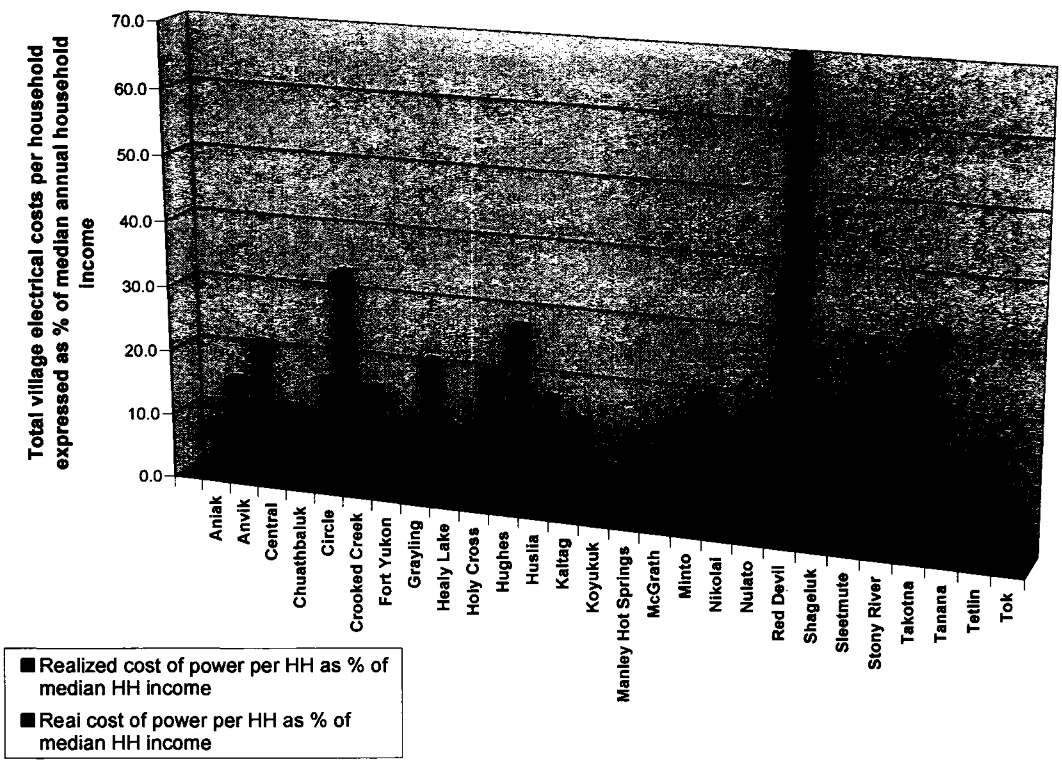

Figure 3.13. Village electrical costs per household. Figures are expressed as a percent of median household income for each community, and include costs incurred for electrical use in private homes as well as in shared facilities such as schools, tribal offices, and washeterias. The discrepancy between the costs borne by consumers and the real cost of power is covered by government funding, primarily via the Power Cost Equalization Program.

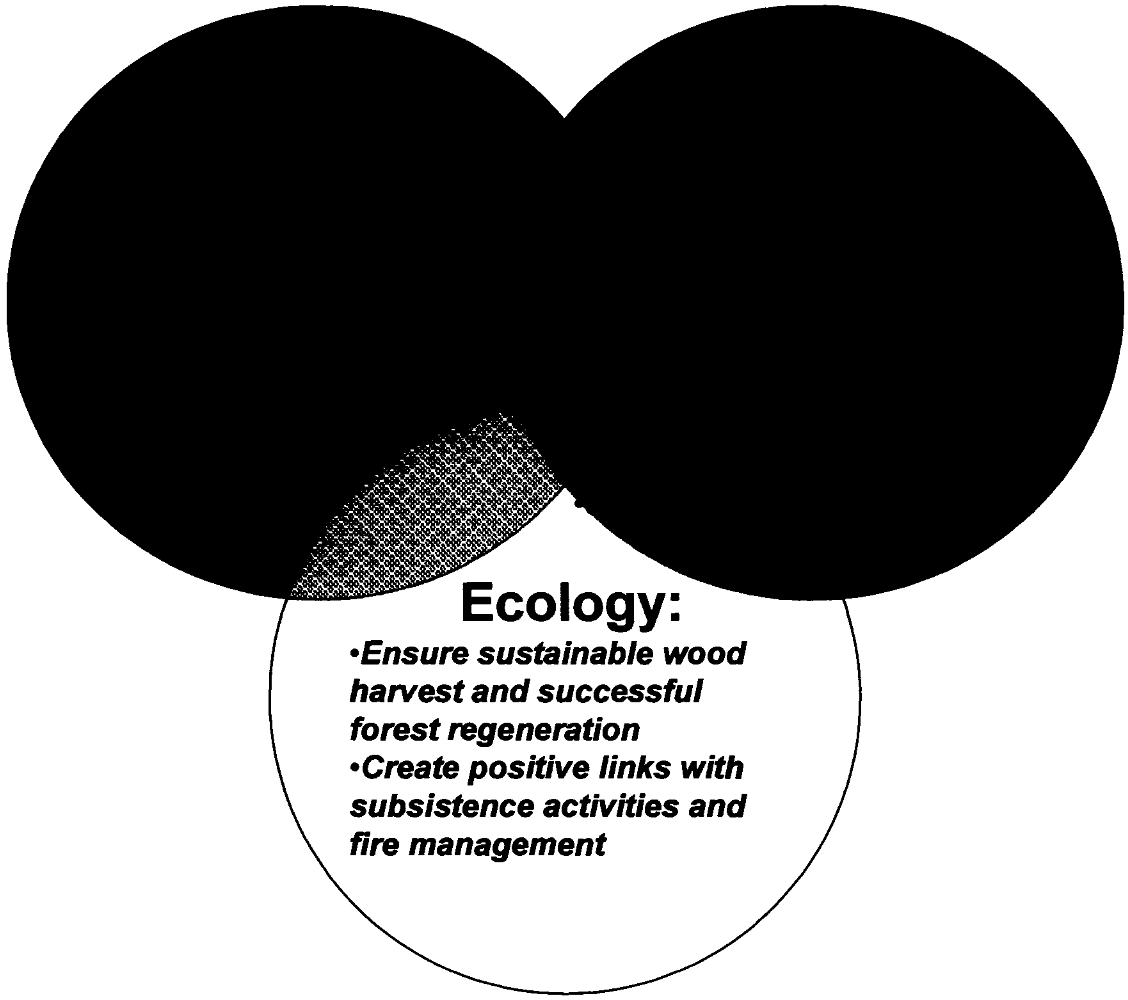

Figure 3.14. Social, economic, and ecological parameters affecting a potential fuel conversion program. These parameters are interconnected and subject to change over time.

Table 3.1. Energy use and costs in forested Interior Alaska communities not on the railbelt electrical grid. The penultimate column indicates what electrical rates would be in each community if Power Cost Equalization subsidies were not provided by the state, and the final column shows the actual rates paid by householders.

Community	Population*	Per capita electrical use (kWh)**	Fuel use (gallons, FY2004)**	Average $/gallon (FY2004)**	Installed generator capacity	Residential rate w/out PCE ($/kWh)**	Actual residential rate w/PCE ($/kWh)**
Alatna & Allakaket	122	5319	53,773	$2.19	430	$0.48	$0.27
Aniak	532	4640	192,576	$1.32	2865	$0.49	$0.32
Anvik	101	4644	38,474	$1.32	337	$0.46	$0.28
Beaver	67	4379	31,436	$1.92	137	$0.42	$0.26
Evansville & Bettles	51	13800	58,368	$1.41	650	$0.41	$0.20
Central	102	4921	50,104	$1.22	640	$0.51	$0.28
Chuathbaluk	105	2036	20,200	$1.70	n/a	$0.56	$0.32
Circle	99	3758	34,750	$1.24	200	$0.50	$0.27
Crooked Creek	147	1731	25,258	$1.69	n/a	$0.56	$0.32
Dot Lake	29		n/a	n/a	325	$0.23	$0.17
Eagle & Eagle Village	183	4270	58,474	$1.20	477	$0.41	$0.26
Fort Yukon	594	4781	207,698	$1.66	2400	$0.34	$0.23
Galena	717	13203	724,076	$1.46	6000	$0.25	$0.18
Grayling	182	3235	46,352	$1.52	546	$0.44	$0.28
Healy Lake	34	4500	14,339	$1.25	105	$0.40	$0.24
Holy Cross	206	3437	54,340	$1.51	585	$0.42	$0.27
Hughes	72		37,325	$3.27	323	$0.51	$0.30
Huslia	269	3409	77,648	$1.79	680	$0.46	$0.28
Kaltag	211	3143	57,498	$1.58	573	$0.46	$0.28
Koyukuk	109	3241	20,830	$1.89	244	$0.45	$0.36
Lime Village	34	2920	9,101	$4.44	77	$0.80	$0.56
Manley Hot Springs	73	4029	26,772	$1.14	480	$0.60	$0.36
McGrath	367	8074	221,650	$1.40	2685	$0.43	$0.29
Minto	207	3491	56,366	$1.13	558	$0.40	$0.26
Nikolai	121	3317	38,182	$1.81	362	$0.50	$0.34
Northway & Northway Village	195	8123	121,569	$1.29	1165	$0.43	$0.25
Nulato	320	3590	85,982	$1.59	897	$0.44	$0.28
Red Devil	35	3612	14,490	$1.83	173	$0.56	$0.32
Ruby	190		24,861	$1.76	654	$0.46	$0.33
Shageluk	132	3073	31,506	$1.69	370	$0.46	$0.28
Sleetmute	78	2939	25,314	$1.69	208	$0.56	$0.32
Stony River	54	2156	13,994	$1.69	139	$0.56	$0.32
Takotna	47	5292	28,219	$1.72	297	$0.48	$0.32
Tanana	304	4533	104,270	$1.34	1456	$0.49	$0.31
Tetlin	129	3669	40,782	$1.46	280	$0.47	$0.27
Tok	1439	8700	861,311	$1.25	4960	$0.23	$0.17

*data from ADCED 2005
** data from AEA 2004
*** data from UAA 2003

Table 3.2. The heating value of wood. Values fall in a relatively narrow range, with spruce near the high end. Gross heating value depends primarily on moisture content.

	Typical Dry-Sample Heating Values (in Btus/dry lb.)		
	Low	Average	High
Hardwoods			
Ash, white	8246	8583	8920
Birch, white	8019	8335	8650
Elm	8171	8491	8810
Hickory	8039	8355	8670
Maple	7995	8288	8580
Oak, red	8037	8364	8690
Oak, white	8169	8490	8810
Poplar	8311	8616	8920
Softwoods			
Cedar, white	7780	8090	8400
Hemlock, eastern		8885	
Pine, white	8306	8603	8900
Spruce, spp		8720	
Residues			
Bark Residue		8629	
Wood Residue		8568	
Woody Yard Trimmings		8600	
Construction Residues		8568	
Min	7780	8090	
Max		8885	8920

Moisture Content	Gross Heating Value (btus/dry ib.)	Ratio GHV/oven-dry GHV
oven-dry	8500	1
25%	6375	0.75
30%	5950	0.7
35%	5525	0.65
40%	5100	0.6
45%	4675	0.55
50%	4250	0.5
55%	3825	0.45
60%	3400	0.4

Metric conversion:
1 kWh =3413 btu
1 lb = .4536 kg
1 Kwh/kg = 7524 btu/lb

Adapted from Maker 2004 and Haq 2002

Table 3.3. Electrical and total efficiency of wood-fired systems. Most authors report greater efficiency from gasification systems than from direct combustion.

Type of process	Electrical efficiency	Combined heat and power efficiency	Source
Hot gasification/fuel cell	0.23	0.6	*Osmosun et al.2004*
Downdraft gasification	0.4	0.9	*Zerbin 1984*
gasification		0.7	*Wu et al. 2003*
gasification	0.35		*Willeboer 1998*
gasification/fuel cell	0.24	0.6	*McIlveen-Wright et al.2003*
combustion	0.25		*USDA 2004*
biomass integrated gasification	0.33		*Haq 2002*
gasfication	0.21		*Somashekhar et el. 2000*
combustion	0.2	0.6	*Bain et al 2003*
mean	**0.28**	**0.68**	

Table 3.4. Capital costs and annual operation and maintenance (O&M) costs for biomass systems as compared to diesel generators. Different authors focus on different scales of operation and different technologies.

Estimated installed cost per kW	Low estimate per kW	High estimate per kW	Estimated O&M costs per kWh	Low estimate per kWh	High estimate per kWh	Plant size	Plant type	Location	Source
$1,536	$1,536	$1,536	$0.17	$0.17	$0.17	100,000 kW	BIGGC*	US	Haq 2002
$914	$914	$914	N/A	N/A	N/A	35,000 kW	BIGGC*	Brazil	Waldheim and Carpentieri 2001
under $2000	$2,000	$2,000	$0.12	$0.12	$0.12	up to 15 kW	BIGGC*	US	Anonymous ENR 2001
$1230-$1488	$1,230	$1,488	N/A	N/A	N/A	5,000-10,000 kW	FBC**	US, Finland	Bain et al 1996
$1400-$2000	$1,400	$2,000	$0.09-$0.14	$0.09	$0.14	25,000-150,000 kW	BIGGC*	US	Bain et al. 2003
$1275-$2000	$1,275	$2,000	$0.17	$0.17	$0.17	25,000-150,000 kW	FBC**	US	Bain et al. 2003
$2,000	$2,000	$2,000	$0.06-$0.12	$0.06	$0.12	2,000-25,000 kW	unspecified	US	USDA 2004
$980-$2500	$980	$2,500	$0.15-$0.20	$0.15	$0.20	1,000-110,000 kW	GS*** or FBC**	US	Scahill 2003
$900-$2200	$900	$2,200	$0.15-$0.20	$0.15	$0.20	15-650 kW	BIGGC*	US	Scahill 2003
Mean	**$1,359**	**$1,849**		**$0.13**	**$0.16**				
$800-$1500	***$800***	***$1,500***	***$0.14-$1.04***	***$0.14***	***$1.04***	***>100kW***	***Diesel generators (rural Interior AK)***	***US***	***EIC 2002, AEA 2004***

***BIGGC** = Biomass integrated gasification combined cycle. Wood chips or chunks are heated in an oxygen-limited chamber to a temperature range of 200-280°C until volatile gases including carbon monoxide, hydrogen, and oxygen are released and combusted.

****FBC** = Fluidized bed combustion. Wood chips or chunks are directly combusted with excess air flow that circulates through the fuel bed.

*****GS** = Grate stoker. Wood chips or chunks are combusted in a simple stoker.

Table 3.5. Costs per acre for forest clearing projects in rural Alaska villages. Costs vary depending on how labor-intensive the work is and how the project is managed.

Fuels treatment project site	Type of treatment	Overhead and equipment cost per acre	Wages per acre	Total cost per acre	Cost per metric ton[1]	Operating cost per kWh[2]
Healy Lake[3]	Fire break	$640	$2,560	$3,200	$282	$0.22
Tanacross[3]	Parklike clearing to spacing of ~12'	$800	$3,200	$4,000	$353	$0.27
Delta Junction[4]	Fire break	N/A	N/A	$1,100	$97	$0.07
Stevens Village[3]	Light thinning of spruce understory	$100	$400	$500	$44	$0.03
Fairbanks[5]	Fire break	N/A	N/A	$2,700	$238	$0.18
mean		**$513**	**$2,053**	**$2,300**	**$203**	**$0.16**

[1] Assuming 28t/ha, .405ha/acre

[2] Assuming 5480*0.85= 4658 kWh/t (green weight)

[3] Data from Hanson 2005 Pers. comm.

[4] Data from BLM 2005

[5] Data from Lee 2005, hand-felling method only

Table 3.6. Estimated annual quantity and value of potential carbon offset credits obtainable via fuel substitution in rural Alaska.

	Total liters of diesel fuel (AEA 2004)	Weight of diesel fuel (kg)[1]	Carbon weight (kg)[2]	CO_2 emissions (t)[3]	Value of credits at current CCX prices[4]	Value of credits at current ECX prices[5]
All PCE communities	107,796,786	84,081,493	72,049,266	263,700	$501,031	$6,328,807
Forested PCE communities in Interior AK	13,329,974	10,397,380	8,909,494	32,609	$61,957	$782,610
Per 1,000 gallons of diesel	3785	2,952	2,530	9	$18	$222

[1]Diesel fuel weighs approximately 0.78 kg/l

[2]Diesel fuel is a mixture of hydrocarbons with an average weight ratio of 12 parts carbon to 2 parts

[3]When combusted, each carbon atom combines with two oxygen atoms at weight ration of C/CO_2 = 12/44

[4]2006 vintage, $1.90/t, September 2005 (CCX)

[5]20 €/t = $24/t August 2005 (McCrone 2005)

Table 3.7. Annual funds for rural Alaska energy projects, including loans and grants. All of these funds are managed by the Alaska Energy Authority (AEA).

	Federal	State				Other Funds			
Funded item/activity	Federal Funds (EPA, HUD, CDBG, DOE)	State Approp.	State Revolving Loan*	Alaska Energy Authority Capital Funds**	Denali Commission	Local funds	Unspecified	Total Funding	ref. year
Circuit rider maintenance and emergency response	$100,000	$200,000						$300,000	2001
Utility operator training								n/a	
Rural Power System Upgrades							$2,300,000	$2,300,000	2000
Rural Power Operations	$68,300	$269,600					$2,400,200	$2,738,100	
Tank farm upgrades	$4,900,000	$2,450,000			$15,350,000	$550,000		$23,250,000	2002
Bulk Fuel Revolving Loan Fund			$51,000					$51,000	2003
AEA Power Project Loan Fund			$835,000					$835,000	2003
Power Cost Equalization		$15,617,225						$15,617,225	2004
Energy Cost Reduction Program***					$2,500,000			$2,500,000	2006
Village End Use Efficiency Program***					$722,000			$722,000	2005
Wind Energy Assessment***	$70,000			$37,000	$390,000			$497,000	2005
Wood Energy Development Program***	$84,000			$16,000				$100,000	2005
Energy Efficiency Technical Assistance***	$137,500			$62,500				$200,000	2005
AEA operation and maintenance							$1,067,100	$1,067,100	2005
TOTAL	$5,359,800	$18,536,825	$886,000	$115,500	$18,962,000	$550,000	$5,767,300	$50,177,425	

* These funds are expressed as annual outlays. They are generally expected to be recouped and recirculated, but a
** As of 2002, assets in the AEA fund were worth $800,000,000
*** Part of the energy conservation and alternative energy development program

data adapted from AEA 2005; AEA 2002; AEA 2004, Alaska 2001, Alaska 2002

Table 3.8. Estimated land area and maximum travel distance for sustainable harvest of black spruce for energy generation. Maximum travel distance ranges from 1.1 km to 12.8 km depending on community size.

Community (or communites)	Population	Annual energy use (kWh)	Load offset (biomass generation capacity = mean load) (kWh)	Harvest area around village (ha)	Maximum travel distance (km)
Alatna & Allakaket	122	648,861	389,317	2665	2.9
Aniak	532	2,468,700	1,481,220	10140	5.7
Anvik	101	469,023	281,414	1927	2.5
Beaver	67	293,400	176,040	1205	2.0
Evansville & Betties	51	703,820	422,292	2891	3.0
Central	102	501,896	301,138	2062	2.6
Chuathbaluk	105	213,737	128,242	878	1.7
Circle	99	372,000	223,200	1528	2.2
Crooked Creek	147	254,434	152,660	1045	1.8
Eagle & Eagle Village	183	781,344	468,806	3209	3.2
Fort Yukon	594	2,840,000	1,704,000	11665	6.1
Galena	717	9,466,799	5,680,079	38885	11.1
Grayling	182	588,761	353,257	2418	2.8
Healy Lake	34	152,986	91,792	628	1.4
Holy Cross	206	708,012	424,807	2908	3.0
Huslia	269	916,941	550,165	3766	3.5
Kaltag	211	663,172	397,903	2724	2.9
Koyukuk	109	353,250	211,950	1451	2.1
Lime Village	34	99,263	59,558	408	1.1
Manley Hot Springs	73	294,120	176,472	1208	2.0
McGrath	367	2,963,200	1,777,920	12171	6.2
Minto	207	722,562	433,537	2968	3.1
Nikolai	121	401,400	240,840	1649	2.3
Northway & Northway Village	195	1,583,944	950,366	6506	4.6
Nulato	320	1,148,831	689,299	4719	3.9
Red Devil	35	126,434	75,860	519	1.3
Shageluk	132	405,639	243,383	1666	2.3
Sleetmute	78	229,258	137,555	942	1.7
Stony River	54	116,418	69,851	478	1.2
Takotna	47	248,705	149,223	1022	1.8
Tanana	304	1,378,060	826,836	5660	4.2
Tetlin	129	473,310	283,986	1944	2.5
Tok	1439	12,518,973	7,511,384	51421	12.8

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Table 3.9. Annual savings in O&M costs and total capital investment associated with three different levels of fuel system replacement. O&M costs include fuel procurement and storage.

Community (or communites)	Estimated annual savings in O&M costs			Estimated installed cost of biomass system		
	Biomass capacity = 1/2 mean load	*Biomass capacity =mean load*	*Biomass capacity replaces 100% of current diesel capacity*	*Capacity to meet 50% of mean load*	*Capacity to meet mean load*	*Capacity to replace 100% of existing generation capacity*
Alatna & Allakaket	$11,320	$25,317	$90,828	$68,479	$136,957	$795,070
Aniak	$7,342	$84,547	$569,857	$260,538	$521,076	$5,297,385
Anvik	$146	$11,945	$88,308	$49,499	$98,998	$623,113
Beaver	n/a	n/a	n/a	$30,964	$61,929	$253,313
Evansville & Bettles	-$7,444	-$3,669	$37,616	$74,279	$148,557	$1,201,850
Central	$5,176	$22,618	$124,348	$52,968	$105,937	$1,183,360
Chuathbaluk	$6,150	$16,173	$67,487	$22,557	$45,114	n/a
Circle	$601	$9,562	$66,458	$39,260	$78,519	$369,800
Crooked Creek	$6,615	$16,765	$67,856	$26,852	$53,704	n/a
Dot Lake	n/a	n/a	n/a	n/a	n/a	$600,925
Eagle & Eagle Village	-$12,195	-$5,423	$66,032	$82,460	$164,921	$881,973
Fort Yukon	-$18,945	$7,847	$224,617	$299,724	$599,447	$4,437,600
Galena	n/a	n/a	n/a	$999,093	$1,998,186	$11,094,000
Grayling	$2,865	$19,017	$117,556	$62,136	$124,272	$1,009,554
Healy Lake	$1,120	$6,035	$35,456	$16,146	$32,291	$194,145
Holy Cross	$2,377	$21,266	$138,694	$74,721	$149,442	$1,081,665
Hughes	n/a	n/a	n/a	n/a	n/a	$597,227
Huslla	$16,168	$47,175	$212,345	$96,771	$193,542	$1,257,320
Kaltag	$7,822	$28,313	$143,901	$69,989	$139,978	$1,059,477
Koyukuk	-$6,399	-$7,724	-$1,937	$37,281	$74,562	$451,156
Lime Village	$15,665	$29,749	$86,051	$10,476	$20,952	$142,373
Manley Hot Springs	$2,590	$14,268	$84,346	$31,040	$62,081	$887,520
McGrath	-$21,238	$24,279	$367,925	$312,726	$625,452	$4,964,565
Minto	-$5,593	$9,675	$121,499	$76,257	$152,513	$1,031,742
Nikolai	$4,549	$11,024	$42,875	$42,362	$84,725	$669,338
Northway & Northway Village	-$36,149	-$45,395	-$24,153	$167,164	$334,328	$2,154,085
Nulato	$5,285	$36,648	$228,618	$121,244	$242,487	$1,658,553
Red Devil	$8,855	$20,129	$73,484	$13,343	$26,687	$319,877
Ruby	n/a	n/a	n/a	n/a	n/a	$1,209,246
Shageluk	$3,856	$15,924	$85,696	$42,810	$85,619	$684,130
Sleetmute	$8,465	$19,640	$73,231	$24,195	$48,390	$384,592
Stony River	$8,450	$19,582	$72,926	$12,286	$24,573	$257,011
Takotna	$5,892	$12,228	$40,154	$26,247	$52,495	$549,153
Tanana	-$5,207	$24,803	$231,579	$145,436	$290,871	$2,692,144
Tetlin	-$4,680	-$3,332	$15,961	$49,951	$99,903	$517,720
Tok	-$353,480	-$463,066	-$380,044	$1,321,209	$2,642,418	$9,171,040

Table 3.10. Advantages and disadvantages of top-down vs. bottom-up strategies for implementing a fuel-conversion program.

	Advantages	**Disadvantages**
Federal government	Power to limit carbon emission laws and treaties	Poor understanding of Alaska
State government	Power to create a statewide program	Emphasis on state rather than commmunity needs
Native corporations	Available capital; interest in village investments	Limited to for-profit activities; no statewide mission
Power cooperatives	Technical knowledge; statewide linkages	Commitment to existing diesel infrastructure
Village councils	Understanding of community needs	Lack of economic and human resources

Table 3.11. Potential hurdles and benefits associated with biomass fuels conversion in Interior Alaska.

	Hurdles	Benefits
Economic	Cost of new infrastructure	Wages from fuel gathering
	Cost of biomass harvest	Reduced cost of diesel
	Certification for sustainable wood harvest	Reduced cost of subsidies
		Market value of carbon credits
Social/Political	Political buy-in from agencies and power companies	Health benefits from reduced pollution
	Ensuring local involvement and continuity	Greater autonomy of local communities
Technical/Ecological	Technical challenges of biomass energy generation	Reduced fire risk
		Greater landscape diversity
	Ensuring long-term sustainability of harvest	Creation of diverse wildlife habitat

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APPENDIX

	Community (or communites)	Access	Population*	Electric Utility*	Total Households*
1	*Alatna & Allakaket*	Koyukuk	122	Alaska Power Company	53
2	Aniak	Kuskokwim	532	Aniak Light & Power Company	174
3	Anvik	Yukon	101	AVEC	39
4	Beaver	Yukon	67	Beaver Joint Utilities	31
5	*Evansville & Bettles*	Koyukuk	51	Alaska Power Company	28
6	Central	Minor Road	102	Central Electric, Inc	67
7	Chuathbaluk	Kuskokwim	105	Middle Kuskokwim Electric Cooperative	33
8	Circle	Minor Road	99	Circle Electric Utility	34
9	Crooked Creek	Kuskokwim	147	Middle Kuskokwim Electric Cooperative	38
10	Dot Lake	Major Road	29	Alaska Power Company	10
11	*Eagle & Eagle Village*	Minor Road	183	Alaska Power Company	90
12	Fort Yukon	Yukon	594	Gwitchyaa Zhee Utilities	225
13	Galena	Yukon	717	City of Galena	216
14	Grayling	Yukon	182	AVEC	51
15	Healy Lake	Minor River	34	Alaska Power Company	13
16	Holy Cross	Yukon	206	AVEC	64
17	Hughes	Koyukuk	72	Hughes Power & Light	26
18	Huslia	Koyukuk	269	AVEC	88
19	Kaltag	Yukon	211	AVEC	69
20	Koyukuk	Yukon	109	City of Koyukuk	39
21	Lime Village	Minor river	34	Lime Village Power System	19
22	Manley Hot Springs	Road	73	Manley Utility Company, Inc	36
23	McGrath	Kuskokwim	367	McGrath Light & Power	145
24	Minto	Minor Road	207	AVEC	74
25	Nikolai	Minor River	121	Nikolai Light & Power Utility	40
26	*Northway & Northway Village*	Major Road	195	Alaska Power Company	62
27	Nulato	Yukon	320	AVEC	91
28	Red Devil	Kuskokwim	35	Middle Kuskokwim Electric Cooperative	17
29	Ruby	Yukon	190	City of Ruby	68
30	Shageluk	Minor River	132	AVEC	36
31	Sleetmute	Kuskokwim	78	Middle Kuskokwim Electric Cooperative	33
32	Stony River	Kuskokwim	54	Middle Kuskokwim Electric Cooperative	19
33	Takotna	Kuskokwim	47	Takotna Community Assoc. Utilities	19
34	Tanana	Yukon	304	Tanana Power Company	121
35	Tetlin	Minor Road	129	Alaska Power Company	42
36	Tok	Major Road	1439	Alaska Power Company	534

***data from ADCED 2005**
**** data from AEA 2004; non-fuel expenses for AVEC villages are calculated at the average rate for the cooperative**
***** data from UAA 2003**

	Avg HH Size*	Median HH Income*	Pop 16 and Over*	Unemployed*	fuel use (gallons, FY2004)**	Average price of fuel (2004)**	Fuel costs	Installed Capacity (kW)***	KWh generated (2004)**	Average load (kW)
1	2.30	n/a	89	20	53,773	$2.19	$117,763	430	648,861	74
2	3.29	$41,875	398	35	192,576	$1.32	$254,200	2865	2,468,700	282
3	2.67	$21,250	69	11	38,474	$1.32	$50,786	337	469,023	54
4	2.71	$28,750	86	12	31,436	$1.92	$60,357	137	293,400	33
5	1.82	n/a	66	n/a	58,368	$1.41	$82,299	650	703,820	80
6	2.00	$36,875	113	8	50,104	$1.22	$61,127	640	501,896	57
7	3.61	$34,286	90	3	20,200	$1.70	$34,340	n/a	213,737	24
8	2.94	$11,667	50	6	34,750	$1.24	$43,090	200	372,000	42
9	3.61	$17,500	90	21	25,258	$1.69	$42,686	n/a	254,434	29
10	1.90	$13,750	18	2	n/a	n/a	n/a	325	n/a	n/a
11	2.03	n/a	140	25	58,474	$1.20	$70,169	477	781,344	89
12	2.62	$29,375	449	52	207,698	$1.66	$344,779	2400	2,840,000	324
13	2.83	$61,125	495	32	724,076	$1.46	$1,057,151	6000	9,466,799	1,081
14	3.80	$21,875	105	13	46,352	$1.52	$70,455	546	588,761	67
15	2.85	$51,250	43	5	14,339	$1.25	$17,924	105	152,986	17
16	3.55	$21,875	165	22	54,340	$1.51	$82,053	585	708,012	81
17	3.00	$24,375	50	3	37,325	$3.27	$122,053	323	n/a	n/a
18	3.33	$27,000	188	21	77,648	$1.79	$138,990	680	916,941	105
19	3.33	$29,167	159	29	57,498	$1.58	$90,847	573	663,172	76
20	2.59	$19,375	68	12	20,830	$1.89	$39,369	244	353,250	40
21	1.79	n/a	n/a	n/a	9,101	$4.44	$40,408	77	99,263	11
22	2.00	$29,000	60	4	26,772	$1.14	$30,520	480	294,120	34
23	2.77	$43,056	286	24	221,650	$1.40	$310,310	2685	2,963,200	338
24	3.49	$21,250	179	29	56,366	$1.13	$63,694	558	722,562	82
25	2.50	$15,000	60	11	38,182	$1.81	$69,109	362	401,400	46
26	3.15	n/a	159	19	121,569	$1.29	$156,824	1165	1,583,944	181
27	3.69	$25,114	213	52	85,982	$1.59	$136,711	897	1,148,831	131
28	2.82	$10,938	29	4	14,490	$1.83	$26,517	173	126,434	14
29	2.76	$24,375	119	17	24,861	$1.76	$43,755	654	n/a	n/a
30	3.58	$26,667	76	17	31,506	$1.69	$53,245	370	405,639	46
31	3.03	$15,000	52	8	25,314	$1.69	$42,781	208	229,258	26
32	3.21	$20,714	49	8	13,994	$1.69	$23,650	139	116,418	13
33	2.63	$14,583	29	0	28,219	$1.72	$48,537	297	248705	28
34	2.55	$29,750	210	31	104,270	$1.34	$139,722	1456	1,378,060	157
35	2.79	$12,250	70	15	40,782	$1.46	$59,542	280	473,310	54
36	2.61	$37,941	995	111	861,311	$1.25	$1,076,639	4960	12,518,973	1,429

	Average load/ Installed capacity	per capita KWh	Total non-fuel expenses (2004)**	PCE payments (2004)**	Residential rate w/out PCE ($/kWh)**	Residential rate after subsidy ($/kWh)**	Real cost of power	Real cost per total kWh	Real cost of power per household
1	0.17	5,319	$83,371	$84,787	$0.48	$0.27	$201,134	$0.31	n/a
2	0.10	4,640	$735,336	$168,391	$0.49	$0.32	$989,536	$0.40	$6,120
3	0.16	4,644	$117,256	$47,007	$0.46	$0.28	$168,041	$0.36	$4,442
4	0.24	4,379	n/a	$17,620	$0.42	$0.26	n/a	n/a	n/a
5	0.12	13,800	$74,967	$34,316	$0.41	$0.20	$157,266	$0.22	n/a
6	0.09	4,921	$148,543	$63,922	$0.51	$0.28	$209,670	$0.42	$4,111
7	n/a	2,036	$69,482	$37,319	$0.56	$0.32	$103,822	$0.49	$3,569
8	0.21	3,758	$86,608	$37,593	$0.50	$0.27	$129,698	$0.35	$3,852
9	n/a	1,731	$68,424	$44,743	$0.56	$0.32	$111,110	$0.44	$2,729
10	n/a	n/a	$15,551	$9,751	$0.23	$0.17	n/a	n/a	n/a
11	0.19	4,270	$128,692	$65,932	$0.41	$0.26	$198,861	$0.25	n/a
12	0.14	4,781	$362,638	$142,391	$0.34	$0.23	$707,417	$0.25	$3,120
13	0.18	13,203	n/a	$124,170	$0.25	$0.18	n/a	n/a	n/a
14	0.12	3,235	$147,190	$69,919	$0.44	$0.28	$217,645	$0.37	$4,544
15	0.17	4,500	$43,540	$13,490	$0.40	$0.24	$61,464	$0.40	$5,152
16	0.14	3,437	$177,003	$83,911	$0.42	$0.27	$259,056	$0.37	$4,464
17	n/a	n/a	$38,238	$27,077	$0.51	$0.30	$160,291	n/a	$6,679
18	0.15	3,409	$229,235	$105,966	$0.46	$0.28	$368,225	$0.40	$4,558
19	0.13	3,143	$165,793	$70,921	$0.46	$0.28	$256,640	$0.39	$4,050
20	0.17	3,241	$18,747	$12,804	$0.45	$0.36	$58,116	$0.16	$1,381
21	0.15	2,920	$62,517	$11,556	$0.80	$0.56	$102,925	$1.04	n/a
22	0.07	4,029	$103,826	$34,735	$0.60	$0.36	$134,346	$0.46	$3,681
23	0.13	8,074	$561,359	$162,757	$0.43	$0.29	$871,669	$0.29	$6,579
24	0.15	3,491	$180,641	$77,094	$0.40	$0.26	$244,334	$0.34	$4,119
25	0.13	3,317	$42,004	$47,474	$0.50	$0.34	$111,113	$0.28	$2,296
26	0.16	8,123	$88,293	$85,818	$0.43	$0.25	$245,117	$0.15	n/a
27	0.15	3,590	$287,208	$138,928	$0.44	$0.28	$423,919	$0.37	$4,888
28	0.08	3,612	$68,461	$16,839	$0.56	$0.32	$94,978	$0.75	$7,652
29	n/a	n/a	$15,999	$19,635	$0.46	$0.33	$59,754	n/a	$868
30	0.13	3,073	$101,410	$42,971	$0.46	$0.28	$154,655	$0.38	$4,194
31	0.13	2,939	$69,424	$41,057	$0.56	$0.32	$112,205	$0.49	$4,359
32	0.10	2,156	$69,067	$16,594	$0.56	$0.32	$92,717	$0.80	$5,512
33	0.10	5,292	$33,897	$20,849	$0.48	$0.32	$82,434	$0.33	$4,613
34	0.11	4,533	$326,127	$109,284	$0.49	$0.31	$465,849	$0.34	$3,908
35	0.19	3,669	$36,882	$48,354	$0.47	$0.27	$96,424	$0.20	$2,085
36	0.29	8,700	$671,543	$212,194	$0.23	$0.17	$1,748,182	$0.14	$3,171

	Real cost of power per HH as % of median HH income	Estimated installed cost of biomass system			Annual operating cost of biomass system		
		to meet 50% of mean load at $1849/kW	*to meet mean load at $1849/kW*	*to replace 100% of existing*	*50% of mean load ($0.17/kWh*	*mean load ($0.17/kWh)*	*100% biomass power*
1	n/a	$68,479	$136,957	$795,070	$44,123	$66,184	$110,306
2	14.6	$260,538	$521,076	$5,297,385	$167,872	$251,807	$419,679
3	20.9	$49,499	$98,998	$623,113	$31,894	$47,840	$79,734
4	n/a	$30,964	$61,929	$253,313	$19,951	$29,927	$49,878
5	n/a	$74,279	$148,557	$1,201,850	$47,860	$71,790	$119,649
6	11.1	$52,968	$105,937	$1,183,360	$34,129	$51,193	$85,322
7	10.4	$22,557	$45,114	n/a	$14,534	$21,801	$36,335
8	33.0	$39,260	$78,519	$369,800	$25,296	$37,944	$63,240
9	15.6	$26,852	$53,704	n/a	$17,302	$25,952	$43,254
10	n/a	n/a	n/a	$600,925	n/a	n/a	n/a
11	n/a	$82,460	$164,921	$881,973	$53,131	$79,697	$132,828
12	10.6	$299,724	$599,447	$4,437,600	$193,120	$289,680	$482,800
13	n/a	$999,093	$1,998,186	$11,094,000	$643,742	$965,613	$1,609,356
14	20.8	$62,136	$124,272	$1,009,554	$40,036	$60,054	$100,089
15	10.1	$16,146	$32,291	$194,145	$10,403	$15,605	$26,008
16	20.4	$74,721	$149,442	$1,081,665	$48,145	$72,217	$120,362
17	27.4	n/a	n/a	$597,227	n/a	n/a	n/a
18	16.9	$96,771	$193,542	$1,257,320	$62,352	$93,528	$155,880
19	13.9	$69,989	$139,978	$1,059,477	$45,096	$67,644	$112,739
20	7.1	$37,281	$74,562	$451,156	$24,021	$36,032	$60,053
21	n/a	$10,476	$20,952	$142,373	$6,750	$10,125	$16,875
22	12.7	$31,040	$62,081	$887,520	$20,000	$30,000	$50,000
23	15.3	$312,726	$625,452	$4,964,565	$201,498	$302,246	$503,744
24	19.4	$76,257	$152,513	$1,031,742	$49,134	$73,701	$122,836
25	15.3	$42,362	$84,725	$669,338	$27,295	$40,943	$68,238
26	n/a	$167,164	$334,328	$2,154,085	$107,708	$161,562	$269,270
27	19.5	$121,244	$242,487	$1,658,553	$78,121	$117,181	$195,301
28	70.0	$13,343	$26,687	$319,877	$8,598	$12,896	$21,494
29	3.6	n/a	n/a	$1,209,246	n/a	n/a	n/a
30	15.7	$42,810	$85,619	$684,130	$27,583	$41,375	$68,959
31	29.1	$24,195	$48,390	$384,592	$15,590	$23,384	$38,974
32	26.6	$12,286	$24,573	$257,011	$7,916	$11,875	$19,791
33	31.6	$26,247	$52,495	$549,153	$16,912	$25,368	$42,280
34	13.1	$145,436	$290,871	$2,692,144	$93,708	$140,562	$234,270
35	17.0	$49,951	$99,903	$517,720	$32,185	$48,278	$80,463
36	8.4	$1,321,209	$2,642,418	$9,171,040	$851,290	$1,276,935	$2,128,225

	Annual diesel fuel costs offset			Annual non-fuel costs offset			Estimated annual savings (compared to real costs of	Per capita annual savings, mean load	Years to pay back capital (mean load, no C credits)
	50% of mean load	*mean load*	*100% biomass power*	*50% of mean load*	*mean load*	*100% biomass power*			
1	$47,105	$70,658	$117,763	$8,337	$20,843	$83,371	$25,317	$208	5.4
2	$101,680	$152,520	$254,200	$73,534	$183,834	$735,336	$84,547	$159	6.2
3	$20,314	$30,471	$50,786	$11,726	$29,314	$117,256	$11,945	$118	8.3
4	$24,143	$36,214	$60,357	n/a	n/a	n/a	n/a	n/a	n/a
5	$32,920	$49,379	$82,299	$7,497	$18,742	$74,967	-$3,669	-$72	-40.5
6	$24,451	$36,676	$61,127	$14,854	$37,136	$148,543	$22,618	$222	4.7
7	$13,736	$20,604	$34,340	$6,948	$17,371	$69,482	$16,173	$154	2.8
8	$17,236	$25,854	$43,090	$8,661	$21,652	$86,608	$9,562	$97	8.2
9	$17,074	$25,612	$42,686	$6,842	$17,106	$68,424	$16,765	$114	3.2
10	n/a	n/a	n/a	$1,555	$3,888	$15,551	n/a	n/a	n/a
11	$28,068	$42,101	$70,169	$12,869	$32,173	$128,692	-$5,423	-$30	-30.4
12	$137,911	$206,867	$344,779	$36,264	$90,660	$362,638	$7,847	$13	76.4
13	$422,860	$634,291	$1,057,151	n/a	n/a	n/a	n/a	n/a	n/a
14	$28,182	$42,273	$70,455	$14,719	$36,798	$147,190	$19,017	$104	6.5
15	$7,170	$10,754	$17,924	$4,354	$10,885	$43,540	$6,035	$177	5.4
16	$32,821	$49,232	$82,053	$17,700	$44,251	$177,003	$21,266	$103	7.0
17	n/a	n/a	$122,053	$3,824	$9,560	$38,238	n/a	n/a	n/a
18	$55,596	$83,394	$138,990	$22,924	$57,309	$229,235	$47,175	$175	4.1
19	$36,339	$54,508	$90,847	$16,579	$41,448	$165,793	$28,313	$134	4.9
20	$15,747	$23,621	$39,369	$1,875	$4,687	$18,747	-$7,724	-$71	-9.7
21	$16,163	$24,245	$40,408	$6,252	$15,629	$62,517	$29,749	$875	0.7
22	$12,208	$18,312	$30,520	$10,383	$25,957	$103,826	$14,268	$195	4.4
23	$124,124	$186,186	$310,310	$56,136	$140,340	$561,359	$24,279	$66	25.8
24	$25,477	$38,216	$63,694	$18,064	$45,160	$180,641	$9,675	$47	15.8
25	$27,644	$41,466	$69,109	$4,200	$10,501	$42,004	$11,024	$91	7.7
26	$62,730	$94,094	$156,824	$8,829	$22,073	$88,293	-$45,395	-$233	-7.4
27	$54,685	$82,027	$136,711	$28,721	$71,802	$287,208	$36,648	$115	6.6
28	$10,607	$15,910	$26,517	$6,846	$17,115	$68,461	$20,129	$575	1.3
29	n/a	n/a	$43,755	$1,600	$4,000	$15,999	n/a	n/a	n/a
30	$21,298	$31,947	$53,245	$10,141	$25,352	$101,410	$15,924	$121	5.4
31	$17,112	$25,668	$42,781	$6,942	$17,356	$69,424	$19,640	$252	2.5
32	$9,460	$14,190	$23,650	$6,907	$17,267	$69,067	$19,582	$363	1.3
33	$19,415	$29,122	$48,537	$3,390	$8,474	$33,897	$12,228	$260	4.3
34	$55,889	$83,833	$139,722	$32,613	$81,532	$326,127	$24,803	$82	11.7
35	$23,817	$35,725	$59,542	$3,688	$9,221	$36,882	-$3,332	-$26	-30.0
36	$430,656	$645,983	$1,076,639	$67,154	$167,886	$671,543	-$463,066	-$322	-5.7

	Years to pay back capital (mean load, CCX)	Years to pay back capital (mean load, ECX)	Estimated annual savings (compared to	Years to pay back capital (100% load, no C credits)	Years to pay back capital (100% load CCX)	Years to pay back capital (100% load, ECX)	Estimated annual savings (compared to real costs of
1	5.3	4.3	$90,828	8.8	8.7	7.8	$11,320
2	6.0	4.8	$569,857	9.3	9.2	8.7	$7,342
3	8.0	6.0	$88,308	7.1	7.0	6.5	$146
4	n/a	n/a	n/a	n/a	n/a	n/a	n/a
5	-47.8	43.6	$37,616	32.0	31.2	24.3	-$7,444
6	4.6	3.7	$124,348	9.5	9.5	8.8	$5,176
7	2.8	2.4	$67,487	n/a	n/a	n/a	$6,150
8	7.9	5.7	$66,458	5.6	5.5	5.0	$601
9	3.2	2.7	$67,856	n/a	n/a	n/a	$6,615
10	n/a	n/a	n/a	n/a	n/a	n/a	n/a
11	-33.9	99.1	$66,032	13.4	13.2	11.3	-$12,195
12	60.9	18.2	$224,617	19.8	19.5	16.6	-$18,945
13	n/a	n/a	n/a	n/a	n/a	n/a	n/a
14	6.4	5.0	$117,556	8.6	8.5	8.0	$2,865
15	5.2	4.2	$35,456	5.5	5.4	5.1	$1,120
16	6.9	5.4	$138,694	7.8	7.8	7.2	$2,377
17	n/a	n/a	n/a	n/a	n/a	n/a	n/a
18	4.0	3.4	$212,345	5.9	5.9	5.5	$16,168
19	4.8	4.0	$143,901	7.4	7.3	6.8	$7,822
20	-9.9	-14.3	-$1,937	-232.9	-281.4	198.7	-$6,399
21	0.7	0.7	$86,051	1.7	1.7	1.6	$15,665
22	4.3	3.5	$84,346	10.5	10.5	9.9	$2,590
23	23.7	12.2	$367,925	13.5	13.4	12.0	-$21,238
24	14.9	9.2	$121,499	8.5	8.4	7.8	-$5,593
25	7.4	5.4	$42,875	15.6	15.4	13.2	$4,549
26	-7.6	-10.9	-$24,153	-89.2	-97.0	5338.9	-$36,149
27	6.5	5.2	$228,618	7.3	7.2	6.7	$5,285
28	1.3	1.2	$73,484	4.4	4.3	4.2	$8,855
29	n/a	n/a	n/a	n/a	n/a	n/a	n/a
30	5.3	4.3	$85,696	8.0	7.9	7.4	$3,856
31	2.4	2.1	$73,231	5.3	5.2	4.9	$8,465
32	1.2	1.2	$72,926	3.5	3.5	3.4	$8,450
33	4.2	3.4	$40,154	13.7	13.5	12.0	$5,892
34	11.3	7.8	$231,579	11.6	11.5	10.7	-$5,207
35	-34.0	62.0	$15,961	32.4	31.2	21.4	-$4,680
36	-5.8	-7.4	-$380,044	-24.1	-25.0	-44.5	-$353,480

	Years to pay back capital (1/2 mean load, no C credits)	Years to pay back capital (1/2 mean load CCX)	Years to pay back capital (1/2 mean load ECX)	Potential annual C credits	
				current CCX prices	*current ECX prices*
1	6.0	5.9	4.4	$860	$10,862
2	35.5	30.4	11.4	$3,081	$38,900
3	338.4	126.1	15.2	$616	$7,772
4	n/a	n/a	n/a	$503	$6,350
5	-10.0	-10.5	-27.2	$934	$11,790
6	10.2	9.6	5.7	$802	$10,121
7	3.7	3.6	2.9	$323	$4,080
8	65.3	47.7	11.5	$556	$7,020
9	4.1	4.0	3.1	$404	$5,102
10	n/a	n/a	n/a	n/a	n/a
11	-6.8	-7.0	-11.0	$936	$11,812
12	-15.8	-17.0	-138.6	$3,323	$41,955
13	n/a	n/a	n/a	$11,585	$146,263
14	21.7	19.7	9.4	$742	$9,363
15	14.4	13.3	7.1	$229	$2,896
16	31.4	27.4	11.0	$869	$10,977
17	n/a	n/a	n/a	$597	$7,540
18	6.0	5.8	4.3	$1,242	$15,685
19	8.9	8.5	5.6	$920	$11,615
20	-5.8	-6.0	-7.9	$333	$4,208
21	0.7	0.7	0.6	$146	$1,838
22	12.0	11.2	6.5	$428	$5,408
23	-14.7	-15.8	-94.0	$3,546	$44,773
24	-13.6	-14.6	-73.4	$902	$11,386
25	9.3	8.8	5.5	$611	$7,713
26	-4.6	-4.7	-6.3	$1,945	$24,557
27	22.9	20.8	9.9	$1,376	$17,368
28	1.5	1.5	1.3	$232	$2,927
29	n/a	n/a	n/a	$398	$5,022
30	11.1	10.6	6.7	$504	$6,364
31	2.9	2.8	2.3	$405	$5,113
32	1.5	1.4	1.3	$224	$2,827
33	4.5	4.3	3.2	$452	$5,700
34	-27.9	-32.0	45.2	$1,668	$21,063
35	-10.7	-11.3	-36.1	$653	$8,238
36	-3.7	-3.8	-4.7	$13,781	$173,985

CHAPTER 4

FOREST CARBON AND SOCIAL-ECOLOGICAL SYSTEMS IN INTERIOR ALASKA: PLACING HISTORICAL, LANDSCAPE-LEVEL, AND LOCAL ANALYSES INTO A RESILIENCE CONTEXT

ABSTRACT

Northern ecosystems and the people who rely upon them are facing a time of unprecedented rapid change due to anthropogenic climate change and cultural and economic pressures. Global boreal forests will play an important role in the feedback loop between climate, ecosystems, and society. In this analysis, I examine forest carbon management in Interior boreal Alaska in the context of social-ecological resilience. I first create a framework in which to assess components of system resilience, and then examine how these components are likely to be affected by regional history, management infrastructure, climate change, and policy responses to climate change. I draw upon an assessment of historical land use and management and current regulatory frameworks; Alaska-based outputs from the Carbon Budget Model for the Canadian Forest Sector (CBM-CFS3); and a case study in carbon management through small-scale local fuel offset programs. My results show system characteristics that tend to bolster resilience as well as features that tend to increase vulnerability. On one hand, low population density, limited fire suppression, and restricted economic expansion in Interior Alaska have resulted in a 21^{st}-century landscape with less compromised human-ecosystem interactions than other boreal regions. Relative wealth and a strong regulatory framework also put Alaska in a position to manage for long-term objectives rather than short-term needs. Moreover, the possibility of successful village-based biomass fuels projects appears promising. However, northern latitudes are likely to be vulnerable to change due to exaggerated climate change and low species diversity, and CBM-CFS3 results indicate that Interior Alaska could switch from being a net carbon sink to being a weak source in the next hundred years, depending on the impacts of ongoing climate change and altered fire cycles. Based on these mixed prospects for resilience, I conclude that land managers

and local communities can potentially bolster social-ecological resilience and help to balance regional carbon dynamics through integrated management of fire, forestry, subsistence, and local energy generation. I argue that in order to reduce vulnerability to rapid change, management goals for Alaska's boreal forest must be long-term, flexible, cooperative, and locally integrated.

INTRODUCTION

The boreal forest of Interior Alaska (Figure 4.1) is facing a set of interconnected ecological, economic, and social challenges, including shifting weather patterns, rising fuel costs, species shifts, and erosion of Native Alaskan traditions and communities. Each of these challenges is linked directly or indirectly to broader global concerns of climate change, carbon dynamics, development pressure, and cultural erosion. In this thesis I have addressed the possibility of obtaining marketable carbon credits in Interior forests from several perspectives, taking into account regional history, the effects of ongoing climate change, and immediate community needs.

In the first chapter, I compared the history of forest use and management in Interior Alaska to that in boreal forests of Canada, Russia, and Sweden. I concluded that Alaska's relatively short development timeline and strong political infrastructure has resulted in retention of key system components, which may convey an advantage for the effective management of local and global ecosystem services, including sequestration of carbon. In the second chapter I assessed the biological and geophysical feasibility of such management objectives. I modeled carbon dynamics at the landscape level, and analyzed the effects of changing fire cycles driven by climate warming. My results indicated that climate change is likely to reduce net carbon sequestration, but that large-scale fire suppression is unlikely to be a practical means of obtaining marketable carbon credits. Thus, I turned instead to a community-based perspective, and in the third chapter demonstrated that Interior villages might obtain carbon credits while reaping other social, ecological, and economic benefits by converting power generation system from diesel fuel to black spruce.

In this chapter I analyze these results in the context of resilience theory. I first review the basic theoretical components of resilience as described by Walker et al. (2004), then translate these components into real-world properties of social, ecological, political, and economic subsystems. Drawing upon the results of my historical analysis, landscape modeling, and community-based case biofuels case study, I describe the degree to which these properties confer resilience or vulnerability in subsystems of Interior Alaska's boreal forest. I also predict what policy directions might tend to increase resilience in Alaska and other circumboreal regions.

A RESILIENCE FRAMEWORK

Resilience theory provides a more useful model than either sustainable development theory or conservation theory for assessing the health of social-ecological systems and for planning boreal forest management, because it recognizes the importance of fundamental system processes, the inevitability of change, and the role of humans within the environment rather than apart from it (Holling et al. 2002; Chapin et al. 2004a). Rural Alaskan communities can be viewed as social-ecological systems whose resilience depends on effective integration of social and ecological functioning in a pragmatic economic context.

Resilience can be defined as the magnitude of disturbance that an ecosystem can weather without shifting to a fundamentally different state (Holling et al.2002). For the purposes of this paper, I rely on the theoretical resilience framework described by Walker et al. (2004). Using the metaphor of a resilience landscape (Figure 4.2), we can imagine non-catastrophic perturbations as movements within the depression currently occupied by the ball. Any disturbance that causes the ball to cross over the edge of a depression will result in fundamental change, although in some cases a new stable state will be reached and in others it will not. For example, shifts between areas a, b, and c are all reversible, and several similar system states are possible within area c, but a shift to area d cannot be reversed. Within a resilience framework, all ecosystems can be seen as being in constant transition due to patterns of disturbance as well as long-term global change (Holling et al. 2002). In other words, the ball is always in motion.

System resilience can be increased (and vulnerability thereby decreased) in several ways. First, resistance to change can become greater, effectively altering the resilience landscape and deepening the depression in which the ball is located. Second, latitude—the degree to which a system can change without crossing a threshold—can increase. This is analogous to widening the depression. Third, the system's position on the landscape can be altered, moving it away from the edges and towards the center. A fourth means of increasing resilience not described by Walker et al. is decreasing system volatility: ongoing perturbations can be dampened or avoided, lessening the chance that any one disturbance will shift the system over an edge. Finally, because social-ecological systems are impacted by effects at multiple scales from the local to the global, resilience at one scale can be subject to unexpected jolts if sudden change occurs at a different scale (Holling et al. 2002; Walker et al. 2004). Throughout this paper I refer to these five qualities as consistency, flexibility, stability, non-volatility and connectivity, respectively.

Boreal Alaska provides an excellent case study of social-ecological resilience because the system has both resilient and less resilient components and regions, and because the current pressures on the system may, under human management, result in either drastic or subtle change. Using the general framework provided by Walker et al. (2004), I create a real-world matrix for judging social, ecological, political/managerial and economic components of resilience (Table 4.1). Focusing on forest carbon dynamics, I then pinpoint Interior Alaska's strengths and weaknesses within this matrix. This approach allows us to broach the question of local resilience while recognizing the linkages that connect Alaska's boreal forest to the rest of the globe.

Social Resilience

At the community level, small Interior Alaska villages with mostly Alaska Native residents are struggling to maintain cultural traditions and viable populations in the face of increasing social and economic pressures (Kerttula 2003; Demarban 2006a). In general, these communities are resilient to change that is slow, predictable, and internal, but vulnerable to change that is rapid, unpredictable, and external (Forbes et al. 2004), such as volatility in fossil fuel markets.

One of the most pressing problems for remote Alaskan villages is that fuel costs (driven by global economic processes) are locally magnified by high transportation costs to rural indigenous communities (Colt et al.2003). Dependence on fossil fuels threatens resilience by increasing volatility (fuel prices are unpredictable); reducing consistency (imported fuels don't foster tradition); decreasing stability (inability to pay for fuel can lead to emigration), and reducing flexibility (if no alternatives are in use).

When fuel costs exceed the capacity of communities to pay, communities must either reduce their dependence on these imports or cease to be viable entities (Demarban 2006a). These villages are the core of an indigenous cultural tradition that depends on ties to the land. If economic factors forced the abandonment of these communities, the associated cultural loss would be large and irreversible. In assessing the problem of high costs for energy and other services, researchers from the Institute of Social and Economic Research note that sustainability is not merely a question of economics, but also of cultural survival. Therefore, the development of community capacity and self-governance– imparting long-term resilience to energy services -- may be more important than the mere existence of these services (Colt et al. 2003). This is similar to the social and cultural choices faced by communities throughout the developing world.

A second problem facing rural communities in Interior Alaska is the question of how to address wildfire. The Alaskan boreal forest is a fire-driven system, with large fires acting as the primary form of disturbance. A complex interaction exists between fire, wages from firefighting, short-term risks from fire near communities, and long-term ecological effects of fire on the landscape.

Many rural residents rely on seasonal wages from firefighting jobs; such jobs bring cash into communities with few sources of external income (Chapin et al 2004c). Thus, fire suppression increases community resilience by creating jobs that improve stabilization, flexibility, and connectivity (through collaboration with state and federal fire managers). Moreover, remote villages are often located in regions with limited fire suppression, and loss of life or property to fire is a real threat during spring and summer months. Reducing this risk clearly represents movement away from a social threshold.

However, fire suppression tends to reduce community resilience in other ways. Modeling results indicate that long-term fire suppression tends to lead to a landscape that is more flammable, due to a higher percentage of mature black spruce (*Picea mariana*) as compared to early-succession forbs, shrubs, and deciduous species (Chapin et al. 2003). Moreover, the natural patchiness of post-fire successional landscapes allows for a mix of early-, mid- and late-succession species, increasing ecological flexibility. This in turn helps foster social consistency by supporting subsistence lifestyles, since the post-fire landscape provides appropriate habitat for a range of animal and plants species upon which traditional communities depend. Thus, reducing fire tends to push the system towards both social and ecological thresholds.

As shown in the previous chapter, fuel substitution may hold promise for Interior Alaska for several reasons. First, fuel offset credits are not one-time credits; as more fossil fuel use is offset over time, more credits can be earned. Second, biomass energy generation can theoretically be developed on a wide range of scales. Finally, and perhaps most importantly, fuel offsets may be possible within a framework that increases the overall resilience of the socio-ecological system in multiple ways.

Increased community autonomy and decreased dependence on imported fuel would reduce perturbations resulting from the inherent volatility of fuel prices. This shift would also be likely to boost the flexibility inherent in multiple economies by creating local jobs, and strengthen traditional ties to the land, thus improving system latitude and consistency. Reduced environmental hazards due to fuel oil or diesel leaks and spills and reduced hazard fuel loads in the rural/urban interface would move communities away from thresholds associated with risk and poor quality of life. Increased wildlife habitat in areas where fire suppression has reduced early-successional vegetation would improve harvest possibilities, thus potentially bolstering subsistence traditions (consistency). Finally, decreased pressure to develop Alaskan oil resources might function as a means of improving resilience across scales, since oil development has the potential to reduce resilience at a broader regional or global level. From this perspective, Interior Alaska village communities are in a position to be at the forefront in developing biomass fuels programs and to serve as pilot projects and leaders in a global movement towards rural biomass power.

Ecosystem Resilience

The boreal forest ecosystem of Interior Alaska has features that tend to bolster resilience as well as features that make it vulnerable to catastrophic shifts. Species plasticity and adaptation to fire disturbance tend to make the ecosystem resilient, while sensitivity to climate makes the system vulnerable to regional climatic change.

At first glance, the low species diversity of Interior Alaska would appear to convey low resilience, given that diversity and redundancy increase flexibility in the face of change and are thus a hallmark of more resilient systems (Peterson et al. 1998; Holling and Gunderson 2002). There are, for example, only 7 tree species in the Interior, so loss of even a single species could have profound ecological effects.

However, what the boreal forest lacks in species diversity, it makes up for in species plasticity (which tends to increase consistency) and landscape diversity (which tends to improve flexibility). Diverse microenvironments result from irregular fire disturbance, variability in slope and aspect, and discontinuous permafrost (Apps and Kurz 1993; Chapin et al. 2004). Throughout the boreal zone, coniferous species dominate the landscape, with deciduous species appearing only in wetter warmer microclimates where permafrost does not lie directly below the ground's surface. Niches disparate enough to be occupied by dozens or even hundreds of species in the tropics are occupied by one or two in the far north. Species have been evolutionarily selected for resilience to shifts in temperature and moisture, since wide variations are the norm (Chapin 1983). Such species will tend to be successful under changing conditions.

Fire, the dominant disturbance in the boreal landscape, is highly erratic in its return interval, severity, and scale (Johnson 1992; Johnson et al.1998) and thus also selects for species plasticity and system flexibility, since the forest has evolved to recover under a wide range of post-fire environments. However, at the edge of a species range or under directional environmental change, when the environment shifts to conditions to which a species is less well adapted, species may be less resilient and more vulnerable to novel environmental conditions (Chapin et al. 2004). In other words, where conditions are already close to a threshold, novel ecosystems are predicted to replace existing boreal ecosystems (Gray 2005).

Despite adaptations to variability and cyclic change, northern regions are particularly vulnerable to the effects of climate change because positive feedback loops tend to exacerbate its effects, thus increasing volatility and reducing stability. Alaska's Interior has increased in mean annual temperature by 2 °C in the past forty years (Keyser et al. 2000). Snow and ice reflect solar radiation, but once melting occurs, darker areas of soil and vegetation absorb more heat (Lashof et al. 1997; Chapin 2004). Once warming begins, it affects vegetation in several important ways, primarily through thawing of permafrost, drought, increased wildfire, insect outbreaks, and skewed species composition during regeneration (Juday 1998; Rees and Juday 2002). Warming can lead to increases in total forest growth (net primary production) but can also increase the decomposition rate of organic matter in litter and soils. Some predictions indicate that these changes may currently be balancing one another, leading to no net change in carbon storage (Keyser et al 2000), but other projections place a stronger emphasis on the water limitations that may accompany warming. Juday et al. (1998) predict that the ecosystem may not be resilient to drought stress, which could result in a dramatic shift to a fundamentally different system, transforming forested land into aspen parkland.

Increased volatility and decreased stability associated with climate change may cause insects to become a threat to boreal regions as warming occurs, either because the insect species have previously been cold-limited and can greatly expand their range and population size under warmer conditions, or because trees that are stressed by changes in temperature and precipitation become more susceptible to infestation (Volney and Fleming 2000). Insect damage in Interior Alaska may result either from increased outbreaks of existing defoliators and boring insects, or from the appearance of new species (Juday 1998; Werner et al.2006).

Boreal forests are not only affected by climate change, but also play an important role in the global carbon dynamics that drive it, underscoring the importance of cross-scale linkages. Anthropogenic increases in atmospheric CO_2 are driven by the release of over 6.5 million metric tons (Gt) of carbon per year from fossil fuels, with an additional 1.6 Gt per year being released by deforestation (IGBP 1998; Innes and Peterson 2001). Currently, approximately 2 Gt of this net flux are being absorbed by carbon sinks (IGBP 1998). The nature of this terrestrial sequestration is uncertain, although it is hypothesized that boreal forests may be an important component of this terrestrial sink (House et al. 2003).

Previous studies have generated mixed results regarding the predicted carbon dynamics of Interior Alaska forests under ongoing climate change. Some suggest that warming might result in modest increases in the sequestration potential of the boreal forest in Alaska and Canada (Yarie and Billings 2002), but others suggest that higher mean annual temperatures might trigger the release of substantial quantities of carbon through increased decomposition and soil respiration, which in turn might affect the rate and magnitude of climatic change (McGuire et al. 2000). Alternatively, sequestration might be reduced by increased fire frequency (Rupp et al. 2002).

As demonstrated in the second chapter of this thesis, results of landscape-level simulations of carbon dynamics in Interior Alaska under three different climate-induced fire regimes using the Alaska Frame Based Ecosystem Code (ALFRESCO) (Rupp et al. 2002) and the Carbon Budget Model for the Canadian Forest Sector (CBM-CFS) (Kurz and Apps 2006) show that while marginal gains are possible, decreases in sequestration could also occur. Increasing mean annual temperatures by 1.5°C or by 4°C resulted in more fire on the landscape, as simulated by ALFRESCO. Such increases in natural disturbances can be expected to reduce resilience by increasing volatility. Increased fire, in turn, resulted in greater carbon losses to the atmosphere as predicted by CBM-CFS.

Since excess carbon leads to positive feedback to climate change, emissions increases act to reduce cross-scale resilience.

In general, large-scale anthropogenic landscape transformation tends to decrease resilience by shifting systems towards ecological thresholds. Increased duration of timber rotations has been suggested as a means of increasing sequestration in the Canadian boreal (Alig 2001), but most of boreal Alaska is not under active timber management, and it is unclear that instituting an aggressive harvest regime at the landscape level would boost system resilience, given that the opposite has been true elsewhere. An increase in average stand age across the landscape could also be attempted through fire suppression. However, such as effort might also have negative impacts on resilience, since fire suppression would tend to increase the proportion of flammable fuels on the landscape (Chapin et al 2003). This, in turn, might result in larger, hotter fires, although this would also depend on precipitation and other factors. Disturbances outside the historic range are an example of increased ecological volatility. Not only would more severe fires pose a greater threat to human life and property, but they would also tend to release more carbon than less intense fires, due to loss of the organic soil layer (Conard et al. 2002; Zhuang et al. 2002). Thus, there is currently little evidence that landscape-level manipulations would be likely to improve the overall social-ecological resilience of the system.

Socio-Political Resilience

In the socio-political realm, vulnerability to catastrophic change can result from lack of effective communication among scientists, managers, policy-makers, and the public; lack of collaboration between control agencies with overlapping goals, resources, land areas, or populations served; application of a rigid ruled-based management structure, as opposed to adaptive management; lack of shared goals between different levels of governance; a history of management shifts that has led to loss of resilience in the social-ecological system being governed; or poor infrastructure and power of enforcement.

Communication is closely linked with public education and dissemination of scientific understanding, and helps foster resilience through non-volatility by avoiding management based on perceived catastrophe. For example, some local residents, visitors, and elected officials questioned why fire suppression efforts in Interior Alaska were not greater during the summer of 2004, when a total of 2.72 million hectares burned, roads were closed, homes were evacuated, and many communities were threatened by fire or densely shrouded in smoke (AFS 2004; D'Oro 2004). However, ecologists raise opposite concerns about the long-term effects of fighting fires, since some fire models predict that suppression will increase the long-term risk of catastrophic fire by allowing the percentage of flammable vegetation to increase (Chapin et al 2003). Fire plays such an important role that understanding and mimicking its effects is seen as a key component of sustainable development of the boreal (Uuttera et al. 1996; Johnson et al. 1998). However, logging, other land clearing, and even controlled burning often fail to mimic the effects of natural fire in ways that are crucial to ecosystem health and resilience (Rees and Juday 2002; Burton et al. 2003; Bergeron et al 2004).

Collaboration between control agencies tends to broaden management horizons, mend cultural and political rifts, and internalize existing economic externalities. Thus collaboration boosts resilience by promoting flexibility and stability, and promoting consistency across political tenures. Conversely, lack of collaboration can increase vulnerability. For example, state foresters from the Alaska Department of Natural Resources (ADNR) are under pressure to create jobs and economic turnover by selling timber. Within the same forest matrix, wildlife managers from the Alaska Department of Fish and Game (ADF&G) are tasked with improving habitat for game species, often by creating areas of early-succession species such as willow and aspen that are preferred over conifers as browse for moose. ADF&G shares neither its funding nor its mission with ADNR. Federal and private land managers add further sets of overlapping and sometimes contradictory directives and guidelines to the mix. Meanwhile, firefighting costs are borne entirely separately from the land management costs associated with recreation, wildlife, subsistence, mining, and timber harvest, despite accruing in the same ecosystem. Finally, Interior Alaska is now demonstrating a political interest in the carbon sequestration potential of Alaska's boreal forest. A state statute in passed in 2004 directs the commissioner of ADNR to recommend policies or programs to enhance the ability of the state to participate in systems of carbon trading (Berkowitz 2004). However, no clear pathway exists for synergy or collaboration between this directive and the others already in place.

Adaptive management tends to increase resilience by fostering novelty and experiment, which in turn helps to create the diversity of options necessary for more rapid and effective response to unexpected change (Holling and Meffe 1996; Holling and Gunderson 2002). Both flexibility and non-volatility can be difficult to achieve within existing political structures. For example, events such as the extreme fire seasons of 2004 and 2005 or the outbreak of bark beetles on the Kenai Peninsula tend to be treated as catastrophic emergencies, rather than as part of an emerging range of forest disturbances that can be attributed to ongoing climate change.

Political resilience is increased when different levels of governance – local, state, and federal – share the same objectives with regard to social-ecological systems. However, this is not always the case in Alaska. Conservation goals tend to be strongest at the federal level, as exemplified by management of designated Wilderness, National Parks, and National Wildlife Refuges, while the state tends to focus on extractive industry. Meanwhile, mitigation of the effects of global climate change requires an even broader level of governance. In 1997, more than 160 nations drafted The Kyoto Protocol on Climate Change, a binding agreement under which developed nations agreed to limit their net release of greenhouse gases relative to 1990 levels (UN 1997). Under the Kyoto Protocol, each signatory nation is required to reduce net emissions to a specified percentage of that country's 1990 emission levels (UN 1997). Article 3 of the agreement allows nations to use verified increases in carbon uptake and storage as credits against carbon release (Wilman and Mahendrarajah 2002; IGBP 1998). Furthermore, Article 6 allows signatory nations to trade or purchase carbon credits with other signatory nations (UN 1997). Carbon management has become a crucial trans-boundary issue, and old notions of sovereignty are becoming outdated (Miller 1998; Ambrose-Oji et al. 2002). Instead, solutions require international cooperation (Breitmeier 1997; Miller 1998). However, institutions able to transcend older patterns and deal with this level of cooperation are still in their infancy.

As demonstrated in the first chapter of this volume, in most boreal regions managers are constrained by systems that have lost social and ecological resilience over the course of centuries, and institutional frameworks left over from the days of maximum sustained yield. Typically, forest management shifted gradually from open access to exclusive control to maximum sustained yield before reaching a stage in which conservation of a broader range of ecosystem services was sought. In each successive stage, land managers tended to consider a broader spatial and temporal scale when weighing costs and benefits. However, only very recently have these scales been broadened to include multi-generational benefits and global impacts. International concerns over anthropogenic climate change have helped to trigger such cross-scale planning, and have thus increased connectivity.

Several regions, including much of the boreal forest in Europe, are faced with ecological systems that have already been shifted close to or across resilience thresholds. For example, Scotland, which was deforested centuries ago, now hosts primarily grassland and shrubland systems (Birks 1988; Tipping 1994). Rural communities depend on grazing rather than forest ecosystem services, and reversion to boreal forests seems ecologically and socially improbable (Angelstam 1998). In Sweden, a less dramatic ecological shift has occurred, although social shifts away from rural subsistence use of the forest are likely to be permanent. Intensive silviculture has boosted wood and pulp production at the expense of indigenous lifestyles and biodiversity (Östlund et al. 1997; Nordlind and Östlund 2003). As many as 6,000 species of invertebrates -- three quarters of the original number -- may have been lost due to reduced forest complexity and habitat destruction (Gawthrop 1999), greatly reducing system stability. Fire has been almost completely removed from the landscape, further reducing diversity, and thus flexibility. Single-aged single-species stands with few snags or wetlands lack a patchy post-fire landscape's resilience to disease, invasive weeds, and climate change (Linder 1998, 2002; Nordlind and Östlund 2003).

In Canada, historical precedent has led to a sometimes-uneasy balance between ecological resilience and social resilience. Although most of Canada's boreal forest has not been transformed to an alternate state through fire control, harvest, or other means, the connection between humans and the land has been pushed towards a threshold by a land tenure system that leases the boreal forest to timber companies. In Grassy Narrows Ontario -- a community that has been previously suffered contamination by mercury from a pulp mill, ongoing flooding of sacred sites, flooding of traditional lands and wild rice fields, and threats to dump nuclear waste on their Customary Lands -- this standoff has been playing out for the past 6 years in the form of a timber blockade by local residents (Carter 2006). Such threats to social traditions reduce social-ecological consistency.

Forest management in Russia serves as an unfortunate example of the importance of cross-scale linkages and stability in fostering resilience. The nation has a forest management framework that is ecologically based and attempts to address concerns at the national and global levels. However, its laws have become unenforceable due to collapse at the local level. Lacking employment or other means of support, members of small rural communities often seek income through unsustainable means. Since the end of the Soviet Union, despite adoption of the Russian Principles of Forest Legislation in 1993 and the Forest Code of 1997, illegal timber trade out of Russia has burgeoned (Gawthrop 1999). Strong management and effective enforcement uphold resilience by fostering stability; the converse is true in this case.

By contrast, as described in the first chapter of this thesis, in Interior Alaska the historical trajectory of land management shifts has been brief enough to preclude protracted damaging phases, and the political and economic infrastructure is strong enough (at least in theory) to take into account local, regional, and global needs and to provide for meaningful enforcement of laws. Many of the changes that have occurred elsewhere have been delayed or avoided. Resilience – as judged by landscape diversity, diversity of ecosystem services, integrity of traditional activities in rural communities, and maintenance of natural disturbance cycles – has been less compromised. Although significant social and ecological change has taken place, the Alaskan boreal forest has not undergone fundamental catastrophic change.

ECONOMIC RESILIENCE

Social-ecological resilience tends to be greatest when traditional economies can successfully be blended with cash economies (increasing flexibility), human and capital investment is diverse and has a long-term outlook (reducing volatility), and full value is given to all ecosystem services (increasing connectivity). These qualities are not necessarily discordant with normative economics, which assumes that all benefit streams can be measured in a common currency, and seeks to maximize these benefits across all stakeholders. In reality, however, several types of market failure tend to lead to sub-optimal results. For example, benefits such as ecosystem services or cultural survival are hard to measure in dollar terms, and tend to be undercounted as a result. Some actions create positive externalities that tend to increase system resilience (e.g. creation of an industry that not only creates a saleable product but also improves wildlife habitat and reduces fire risk) while others reduce resilience (e.g. creation of an industry that damages water quality and fragments habitat). In the world's boreal forests, markets for timber and other forest products now exist at all scales from local to global, increasing the importance of cross-scale linkages such as incorporation of local knowledge and sound science into policy, and fair valuation of ecosystem services at all scales; however, government subsidies and other market manipulations sometimes undermine resilience instead (Table 4.2).

Furthermore, even if a normative equation were easily rectified with the complexities of real-world conditions, maximizing utility would not necessarily be desirable from a resilience perspective because total benefits tend to be greatest when a system is close to a threshold of system collapse. For example, in calculating maximum sustained yield (MSY), the greatest annual benefits will be gained if harvests are set very close to the theoretical maximum growth for the harvested population, but such policies lead to volatility. In addition, once a system has shifted to a less than optimal state, it may be necessary for managers to temporarily reduce total benefits in order to allow the system to rebound (Scheffer et al. 2002).

Markets for non-timber forest goods and services are still nascent or nonexistent, and many of these benefit streams – including atmospheric carbon -- are still treated as public goods. Managing public goods is challenging even within nation-states, and can be extremely difficult outside the bounds of traditional sovereignty (Lipschutz 1998; Ward 1998; Schnaiberg et al. 2002). This is particularly true in the case of a complex linked assembly of resources, some of which can more easily be incorporated into a free market system than others. Hardin's (1968) theory of the tragedy of the commons posits that in order to avoid inevitable degradation, resources must be either privatized or held in common by governing agencies that control and restrict use and access. However, later theorists (Feeny et al. 1990; Acheson 1989; Ostrom 1990; Dietz et al. 2003) see a broader range of options that includes community management, citizen enforcement, and co-management.

The growing international trade in carbon credits offers a link that may help lead to the development of better trans-boundary cooperation, a more encompassing framework for forest management, and increased resilience through improved cross-scale linkages. The transferability of carbon credits has opened up international economic possibilities never before seen, although some parallels can be drawn to the successful mitigation of SO_2 pollution in the US through use of tradable pollution credits (CCX 2006). Some of the world's most developed nations are already taking a lead; in January 2005, the European Union initiated a legally binding international trading market in greenhouse gas emissions (Kirk 2004). Meanwhile, binding agreements have been made and non-governmental markets have already appeared, even in nations that are not signatories to the Kyoto Protocol. For example, in August 2001 the New England Governors and Eastern Canadian Premiers signed a regional climate change agreement aimed at reducing greenhouse gas emissions to 1990 levels by 2010, and reducing emissions by 10 percent below 1990 levels by 2020. Meanwhile, the Chicago Climate

Exchange (CCX) is acting as a self-regulating voluntary market, administering the world's first multi-sector and multi-national emission-trading platform. By participating in trading through CCX, corporations, municipalities, and other institutions have made legally-binding commitments to reduce net emissions of greenhouse gases. Carbon emitters as well as credit holders are banking on the idea that the price of credits will escalate eventually, either due to international agreements or state and local laws. By entering the market early, they are showing good will and environmental responsibility, as well as setting up relationships that may prove lucrative in the future.

From an economic standpoint, forest sequestration can be viewed as a way of deferring negative impacts and thereby lessening total costs to society, assuming the application of a discount rate; it can also be viewed as a renewable rental contract or as a means of buying time while more effective alternative technologies are being developed (Wilman and Mahendrarajah 2002). From a resilience perspective, it offers a way to move away from thresholds (albeit temporarily) while bolstering understanding of cross-scale linkages within management frameworks.

CONCLUSION

Not only is climate change inexorably affecting forests, but forests are also influencing climate change. Thus, the questions central to ecosystem management are also changing. When analyzed in the context of resilience, the results of my historical analysis, landscape modeling, and community-based case biofuels case study indicate that social-ecological vulnerability would be reduced in Interior Alaska if the paradigm of ecological sustainability were replaced by an even newer conceptual framework. In this framework, the time scale and spatial scale applied to forest management decisions is broadened even further and the resilience of forests and their users become paramount.

In Interior Alaska, there are several sound economic, ecological, and social reasons to pursue carbon management as a focal point of boreal forest management. First, initiating carbon accounting is a proactive strategy; it addresses the likelihood that laws regulating carbon releases and sequestration as well as market values for credits will become more important over time. Second, including a global concern such as atmospheric carbon in management decision-making processes necessitates the creation of an inclusive and collaborative management framework that may be more flexible to the future inclusion of other cross-scale issues and previously uncounted ecosystem services. Finally, development and implementation of carbon management strategies may create immediate local benefits, either directly or indirectly, by securing new income streams, increasing local autonomy in certain arenas, and reducing passivity and helplessness in the face of ongoing climate change.

Boreal Alaska is among the most likely regions to have both the societal impetus and the necessary ecological characteristics to be the testing-ground for a profound shift in the focus of forest management and forest conservation. Attributes that are likely to boost social-ecological resilience include a strong regulatory framework, statehood within a wealthy and powerful nation, little history of sequential and degrading management, and a large number of small rural indigenous communities that may potentially benefit from projects that bolster both local and global resilience.

Nevertheless, Interior Alaska faces significant challenges to achieving multi-scale social-ecological resilience. Major hurdles include not only the inherent complexity of the problem, but also short political tenures that make long-term planning difficult; multiple control agencies with sometimes conflicting directives; the existence of non-contiguous areas as management units; existing economic externalities; and ongoing effects of climate change that may shift the system across thresholds regardless of management choices. Moreover, land managers currently face a strong political mandate for natural resource development, including maximization of wildlife harvest, mining oil and gas development, and timber harvest. Many decisions that will potentially affect local resilience will be made outside the state.

Given the above strengths and weaknesses, application of resilience theory emphasizes the importance of flexible management, innovative and cooperative approaches, and integration of local knowledge and local needs with management directives at broader scales, up to and including the global level. Thus, the related challenges of fire management, climate change, ecological shifts, community integrity and cultural survival can be addressed as integrated components of an interdependent system.

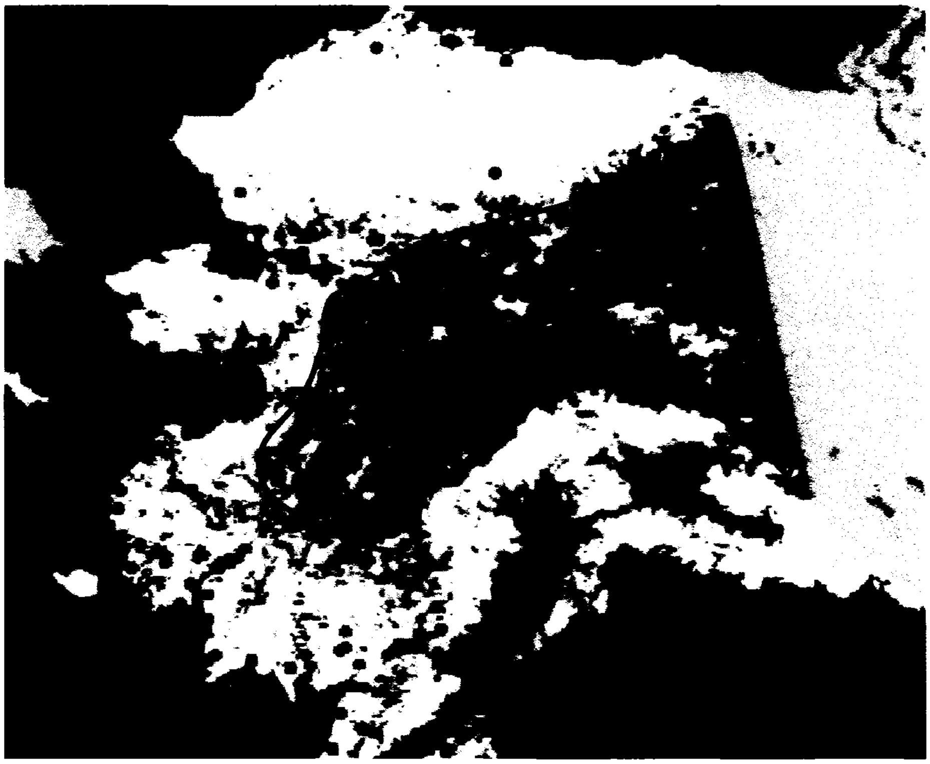

Figure 4.1. Remote Alaska communities in forested Interior Alaska. Approximately 45 remote communities lie in the region considered in this study (roughly demarcated by black line). (adapted from Crimp and Adamian 2000).

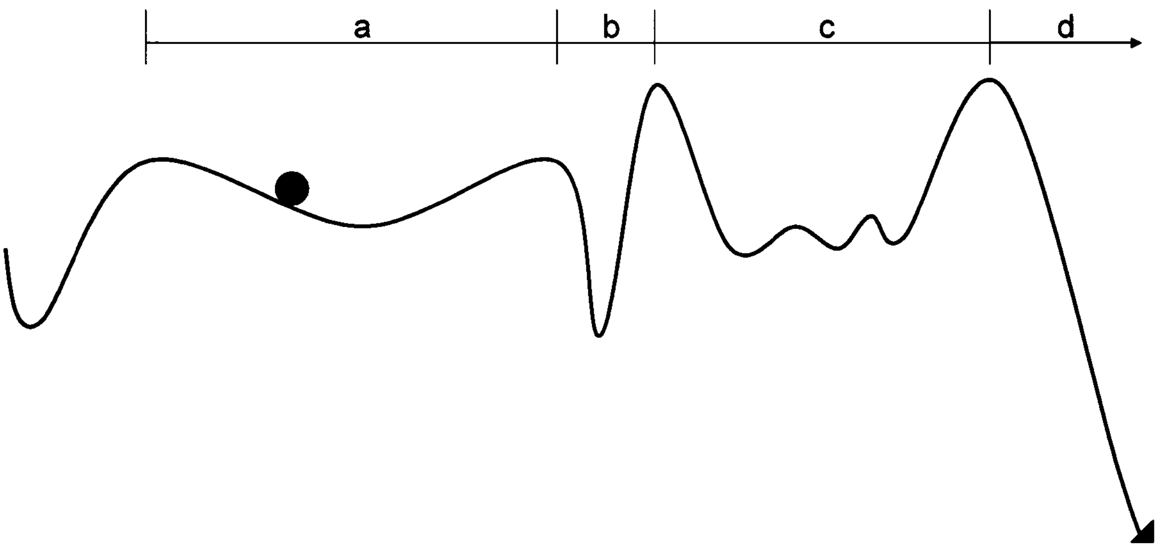

Figure 4.2. Schematic of a resilience landscape. The resilience of the social-ecological system is determined by the contours of the resilience landscape, the current position of the ball, and the magnitude of the ball's movements.

Table 4.1. Factors affecting social-ecological resilience. Overall system resilience depends on attributes of society, ecology, policy, and economics. Vulnerability can be reduced by increasing resistance to change, by improving system flexibility, by shifting the system away from catastrophic thresholds, by reducing the amplitude of perturbations, or by strengthening connections across scales. The stability traits are mostly command-and-control, not system properties.

	Resistance to change (consistency)	**System latitude (flexibility)**	**Movement away from thresholds (stability)**	**Reduced amplitude of perturbations (non-volatility)**	**Cross-scale linkages (connectivity)**
Social	Strong traditions and willingness to defend beliefs	Cultural adaptation and formation of mixed economies	Viable populations and sustainable sources of food and income	Consistent community leadership; reliance on less volatile resources	Good communication between local communities and other land users
Ecological	Species plasticity	Overlapping functions of species; incorporation of new species	Avoidance of large-scale anthropogenic landscape transformation	Natural disturbances within historic range	Refugia outside the region
Political/ managerial	Policies designed to outlive political administrations	Adaptive management strategies and innovative incentives	Strong management agencies and effective enforcement	Policy based on long-term trends, not reaction to perceived catastrophes	Incorporation of local knowledge and scientific research into policy
Economic	Commitment to economies based on lifestyle, not short-term gain	Overlapping economies based on a wide range of resources and skills	Funding and human capital invested in sustainable economies	Modest harvesting; surplus funds invested	Ecosystem services valued at all scales

Table 4.2. Examples of timber subsidies and other market manipulations that tend to lead to market failures in four nations. In most cases, numerous examples are available in each category.

Type of subsidy	Examples by nation			
	United States	*Canada*	*Sweden*	*Russia*
Government financing of road-building or other infrastructure needed for timber and fiber harvest and transport	Federally funded logging roads in the Tongass National Forest (2004): $49 million; timber receipts: $800,000 (Walder 2005)	Although some logging roads are privately constructed, heavy loads of timber are transported hundreds of miles on the public road system	As in Canada, the U.S., and elsewhere, wear and tear by logging trucks on publically funded roads creates a negative economic externalitiy	In the Soviet era, the government hauled wood from east of the Urals to mills in the west, a practice that has collapsed without subsidies (Gawthrop 1999)
Timber harvesting on public land at below market prices and/or at the expense of other public uses	The National Forest Timber Sales Program operates at a net loss of roughly $800 million annually (Hanson 1996)	Timber lease between provinces and private companies undercount non-timber values, and are thus under-valued (Adamowicz 2003, Scarfe 1998)	Below-cost government sales occurred historically; now most forest is privately owned (Burton et al. 2003)	Government determination of prices was the status quo in the non-market economy during the Soviet era (Gawthrop 1999)
Direct payments, reduction of taxes, technical assistance, or waiving of regulations for timber operations	U.S. reports to The World Trade Organization more than $600 million in Federal subsidies to the forest industry through tax concessions (Joshi 1996)	Subsidies in the Canadian softwood market led to 14 years of free trade disputes between the US and Canada (Joshi 1996)	Growers of rare high-value trees receive direct assistance; tax breaks were given to clear damaged timber after a large storm in January 2005 (Lexmon 2005)	Non-collection of taxes from massive illegal logging operations serves as an indirect subsidy (Harman 2005)

CHAPTER 4 LITERATURE CITED

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